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New insights for the future of Lake Champlain: Practical approaches and useful tools for grappling with uncertainty and weighing trade-offs in watershed management.

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NEW INSIGHTS FOR THE FUTURE OF LAKE CHAMPLAIN: PRACTICAL APPROACHES AND USEFUL TOOLS FOR GRAPPLING WITH UNCERTAINTY AND WEIGHING TRADE-OFFS IN WATERSHED MANAGEMENT.

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ABSTRACT

The effective management of non-point source nutrient pollution continues to prove elusive. Though the scientific literature is unequivocal that all anthropogenic land uses contribute to non-point source (NPS) pollution, variable levels of contribution over time and across location and complex relationships between cost and effect make finding technologically effective management solutions difficult. In addition, these solutions are implemented in a world of scarce resources, diverse and often competing concerns and values, and intense public scrutiny. Clearly, making the best possible decision about how to manage NPS pollution under these conditions is not simple. My overarching goal was to develop and test several practical approaches that provide insight into the implications of management decisions and the trade-offs facing water quality managers using the challenges of restoring Lake Champlain as a test case.

I first demonstrate a simple spreadsheet-based method for (1) identifying the areas of greatest potential for further phosphorus reductions, (2) estimating the potential scale of those reductions, and (3) identifying the severe tradeoffs that exist between cost and effectiveness at high levels of management. Results of this method suggest that better and more extensive management of developed impervious surfaces and annual cropland and hayland represent the greatest potential for phosphorus reductions. Farmstead management, combined sewer overflows, and wastewater treatment present little opportunity under the current regulatory environment. Results also suggest that due to order-of-magnitude differences in cost-effectiveness between management practices for developed and agricultural lands, substantial tradeoffs exist between cost-efficiency and equity in the distribution of responsibility for management.

Second, in an effort to quantify the variability of NPS contributions over time and space, I developed and applied a Bayesian hierarchical modeling approach to incorporate annual hydrologic variability and uncertainty about land use areas into estimates of land-use specific phosphorus loading rates and watershed-scale residual loading. The model was able to replicate both average load and the variability around that average with an acceptable degree of precision. The results of this approach suggest that for some watersheds, unmanageable sources of phosphorus are dominant.

Third, I applied a Bayes network to predict the effects of alternative management scenarios on phosphorus loads. Using evolutionary optimization and a multiple-criteria decision analysis, I explored the tradeoffs between cost, effectiveness, and distributional equity in the burden of management. Results of this study indicate that the probability that phosphorus loads will comply with regulatory targets is, in some watersheds, small under any management scenario. More interestingly, it also appears that there are large differences between watersheds in the ability of management actions to raise those probabilities, and the significant and non-linear tradeoffs between cost, effectiveness, and equity will make decision-making – and achieving restoration targets – difficult.

Together, these approaches provide a foundation for a fuller and more completely informed decision-making process that incorporates uncertainty and identifies key trade-offs for the State of Vermont as it implements a new management plan for Lake Champlain.

ACKNOWLEDGEMENTS

The work presented here is the result of a team effort, and without their support, this dissertation would not have the quality that it does, and the process of creating it would not have been nearly so interesting, and – at times – fun. They all deserve recognition.

First, my committee deserves a lot of credit. I did not start out knowing very much about anything I discuss in the following pages, and individually and collectively, their support and consistent availability was vital for my learning process. Many extra thanks go to Mary, my advisor, for granting me the trust and the total freedom to explore ways of thinking that I thought were interesting and useful.

Several individuals around the Lake Champlain basin were invaluable for providing me with data and with insights about the complexities of actually managing watersheds (instead of just *thinking* about managing watersheds). Eric Howe at the Lake Champlain Basin Program, Eric Smeltzer at Vermont DEC, Kip Potter at VT-NRCS, and Eric Perkins at EPA Region 1 were all especially helpful in this regard.

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Introduction

Lake Champlain forms the border between much of western Vermont and northeastern New York. The lake drains through the Richelieu River through Quebec to the north, and its watershed area lies within the jurisdictions of the states of Vermont and New York, and the Canadian province of Quebec. Like many lakes in North America with well-developed watersheds, Lake Champlain suffers from a variety of ecosystem-level pressures. Most notably these pressures include introductions of invasive aquatic vegetation and fish species, loading from diffuse toxic contaminants, and the effects of excessive nutrient loading in the form of phosphorus (LCBP 2003). The latter is evident mostly through summertime algae blooms, which have become increasingly dominated by cyanobacteria over the last two decades (Watzin et al. 2011). This review explores the scope and nature of the phosphorus problem, the current management strategy, and insights from the decision-sciences in relation to areas for improvement.

The Lake Champlain Basin: geography and land use

Lake Champlain is situated between Vermont's Green Mountains to the east, and New York's Adirondacks to the west (Figure 1-1). The majority of the watershed lies within Vermont's borders (56%), while over a third lies in New York (37%) and the remainder (7%) in Quebec. The watershed-to-lake area ratio is large compared to similarly sized lakes (19:1), and the lake drains a total area of 21,356 km^2. The lake is over 190 km long and is only 19 km across at its widest (Stickney et al. 2001). The relatively narrow valley contributes to strong seasonal winds oriented along the valley floor that can slow the movement of water into and out of the many shallow bays along the lake's shoreline. Combined with the presence of over 70 islands and the associated bridges and causeways, these meteorological and geographic factors have contributed to the development of five hydrologically distinct major lake segments, each with a unique combination of water and sediment chemistry and hydraulic residence time.

Immediately following the most recent glacial retreat, the Lake Champlain Basin was inundated twice – first with a large freshwater lake (Glacial Lake Vermont), and second with a shallow inland sea (the Champlain Sea) – which led to heavy deposition of fine-textured soils in both instances. As these two large water bodies drained over time, the silt and clay deposited on the tributary deltas and the lake and sea floors resorted onto a plain that now makes up much of the Basin lowlands. These fine textured soils have proven especially productive for agriculture due to their high nutrient holding capacity and natural enrichment by both the marine deposition of the Champlain Sea and the calcium-rich nature of the bedrock (Johnson 1998).

Land use in the basin has been and continues to be widely varied. During the early 1800s, both Vermont and New York relied heavily on the timber industry. By the mid-19th century the vast majority of the land had been cleared and converted to agricultural production. During this period, the volumes of sediment and water delivered to streams increased greatly due to lack of ground cover, and streams across New York and Vermont aggraded and deposited huge amounts of sediment over their floodplains. Over the next eight decades, wide-scale reforestation (both active and passive) reduced sediment loads to streams, and the channels across the Basin subsequently cut down through the sediments, losing access to their floodplains. In addition to these changes in hydrology, an increase in urban and suburban development in the basin (and the concomitant increase in impervious surface) in the past 50 years has further forced heavy scouring of streams. In short, streams across the basin have experienced widespread and severe change in planform and cross-sectional geometry, with greatest impact on their ability to carry, store, and deposit sediment and nutrients in the channel and on floodplains (Kline and Cahoon 2008).

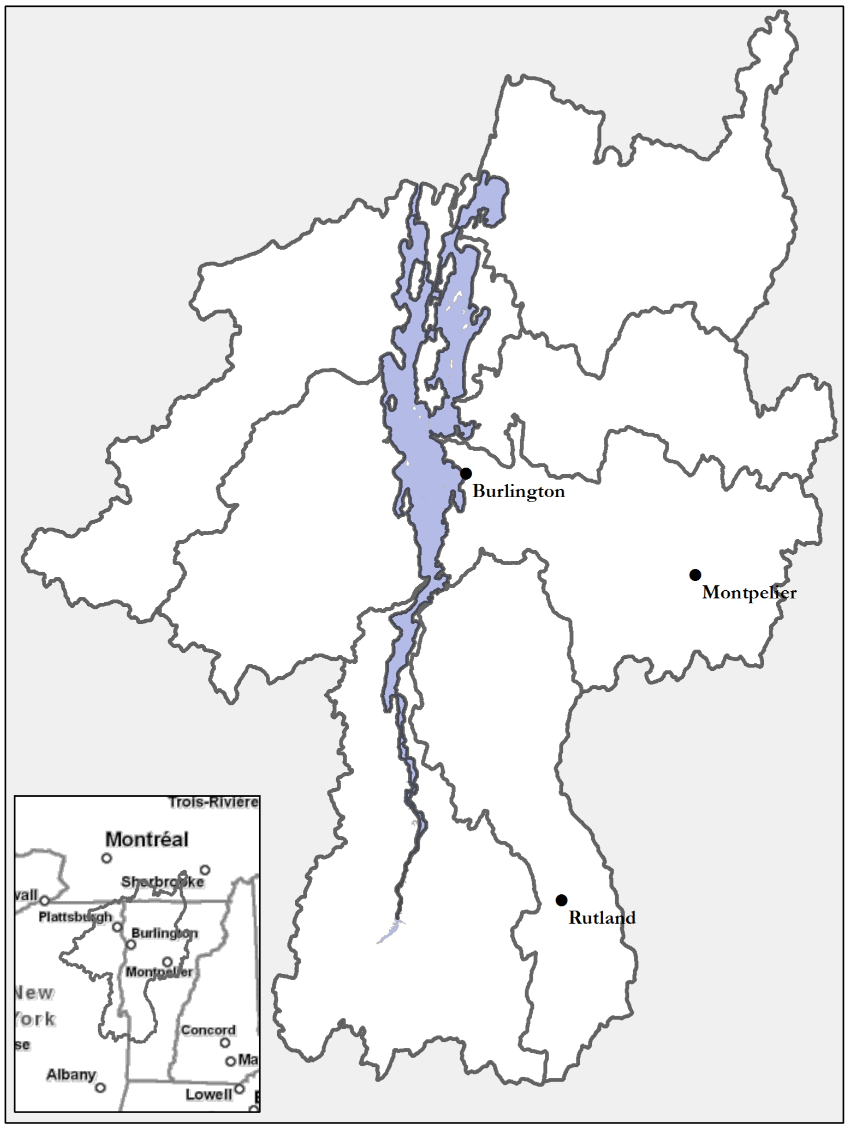

Figure 1-1. The Lake Champlain Basin, showing boundaries of major sub-watersheds

The most recent estimates of land cover in the Lake Champlain Basin (Jin et al. 2013) show that forest predominates current land cover basin-wide (66.4%), while agricultural and urban land uses comprise much of the remainder of current cover, at 14.1% and 6.5%, respectively.

The Development of the Diffuse Phosphorus Problem

Lake Champlain's phosphorus loading problem has its roots in the unique combination of the Basin's recent agricultural activity, current development pressure, and the history of land use and its effect on stream geomorphology. Though forested lands comprise the vast majority of the land area in today's basin, they contribute only an estimated 36.8% of the phosphorus load (Tetra Tech 2013). This is likely close to the natural "background" level of phosphorus loading, and is considered in large part an unmanageable source. Localized areas, however, such as along road cuts through forested areas and where logging activity is intensive, can represent important sources of sediment and phosphorus. Agricultural land, urban land, and erosion from the stream channel constitute the vast majority of the load, and the first two are manageable sources. While the manageability of the third is subject to debate, for the purposes of this review, we consider manageable over the very long term. The development and contributions of all three sources are discussed below.

Agricultural Sources: Fertilizer

Most farms in the Champlain Basin are situated along the valley floor where the dominance of fine-textured soils provides exceptional yields of hay and grain crops. With the exception of a period in the mid-1800's when sheep were more common, farms across the Champlain Basin have historically been predominantly pasture-based cow dairies, growing only small amounts of grain to supplement winter feeds. However, once train refrigeration made access to the urban liquid milk markets of Boston and New York easy, Vermont and northwestern New York farms increasingly purchased grain – primarily wheat and corn – as a feed supplement to maintain high milk production year-round. The development of cheap synthetic fertilizers during the 1930's added to the Lake Champlain Basin's imports of chemical phosphorus; over the next two decades, unrestrained chemical fertilization became increasingly common. Since the 1970's, fertilizer use in Vermont has dropped because of its high price, while manure use has increased, which has had two major effects on the dynamics of soil phosphorus on farms (Ford 2012). Firstly, manure is generally applied to satisfy the nitrogen demand of crops. Dairy manure has a relatively high proportion of phosphorus (particularly dissolved phosphorus) in relation to nitrogen (Jokela et al. 2010). As a consequence, phosphorus is often over-applied (i.e., in excess of crop need) in a typical nitrogen-based nutrient management scheme (Maguire et al. 2008). Secondly, the high organic content of manure acts as a reservoir for labile forms of phosphorus, adsorbing dissolved phosphorus and re-releasing it slowly as the organic matter breaks down. The dairy producers' demands for imported high density feeds and their heavy use of chemical fertilizers over the past 40 years has contributed to soils that have slowly accumulated phosphorus throughout the Champlain Basin (McDowell et al. 2002).

Agricultural Sources: Feed Supplements

Concurrent with the excessive use of synthetic fertilizer, phosphorus was routinely given in excess in mineral dairy supplements (>0.50% phosphorus) to ensure high milk yield and high reproductive capacity, often at the advice of university academics, Extension personnel, and industry nutritionists (Anderson and Magdoff 2000). Not until the 1970's did it begin to become clear that this level of phosphorus feeding was unnecessary, and in fact was a contributor to the build-up of phosphorus in agricultural soils and the eutrophication of surface waters. Rather than being taken up or metabolized, the bulk of phosphorus not needed by the cows is excreted in manure (Cerosaletti et al. 2004), mostly in the dissolved inorganic form (Dou et al. 2002). In the majority of farming systems this manure is eventually spread on crop and hay fields, and the dissolved phosphorus fraction is at high risk for being transported in surface runoff or leached through sub-surface tile drains. In recognition of the adverse effects of feeding excess phosphorus, in 2001 the National Research Council lowered its recommended phosphorus supplement feeding rate for lactating cows from 0.45% to between 0.35% and 0.38% (NRC 2001); despite this fact, dairy cows are still often fed in excess of 0.40% in the Lake Champlain Basin (Contach et al. 2003). Analysis of manure samples collected over the period 1992 to 2006 show a slow reduction in average phosphorus content, possibly indicating a gradual adoption of the lowered phosphorus supplementation rates (Jokela et al. 2010).

Agricultural Sources: Erosion

While fertilizers and feeds have comprised the major imports of phosphorus to farms in the Lake Champlain Basin, erosion of sediment and dissolved phosphorus runoff from storm events are responsible for delivering both the dissolved and particulate phosphorus fractions to streams. Delivery of phosphorus to surface waters and resulting eutrophication occurs where there are both potent *sources* of phosphorus (such as saturated soils) and *transport mechanisms*, such as a strongly sloped field. Most phosphorus in the soil is held in varying degrees of association with soil particulates (i.e., it can be fixed or labile), while typically a much smaller proportion is dissolved in the soil solution (Gburek et al. 2005). Where these sources and transport mechanisms come together, "critical source areas" can contribute disproportionately to phosphorus loading (Sharpley and Tunney 2000).

In the Lake Champlain Basin, agriculture is the most common land cover type after forest, and of the roughly 208,000 acres of harvested cropland in the Vermont portion of the Basin, close to 45,000 of those acres are in corn in any given year[1] (USDA 2009b). According to the same census, the proportion of harvested cropland in corn is slightly higher in the New York portion of the Basin than in Vermont (29.2%, as opposed to 21.9%), which indicates that New York grows about 59,000 acres of corn in its almost 202,000 acres of harvested cropland in the Basin (USDA 2009a). The great majority of these 110,000 acres of corn in both states is harvested for silage, which means that very little of the crop residue is left on the ground at the end of the season. Furthermore, fall plowing to incorporate fall-applied manure is a common practice across New England (typically between mid-October and mid-November), as is plowing just before planting in the spring to incorporate a second manure application and to dry the soil (typically late April or early May).

Precipitation events during this extended period of no plant cover are responsible for a large proportion of the soil erosion that occurs during the entire year. Particularly problematic are those storm events that occur when the ground is frozen, as no infiltration occurs and all precipitation becomes overland flow (Zuzel and Pikul 1987). Without the protection of a plant canopy, a litter layer, and well-established plant roots to hold soil in place, rain and overland flow easily dislodge soil particles and transport them into nearby water bodies (Walling 1999, Smil 2000). During storms, farmsteads can become a particularly potent source of sediment and of dissolved phosphorus because of the coincidence of sources and transport mechanisms. For example, where barnyards are not

[1] These figures are calculated from the Vermont totals from the 2007 US NASS Ag Census Survey, and divided by the proportion of the state lying within the Champlain Basin (48%); 433,074 acres of harvested cropland and 92,771 acres of corn, each multiplied by 0.48 = 207,877 and 44,530 acres, respectively. Additionally, though producers sometimes rotate corn in and out of a given field, in Vermont, the total annual acreage of corn is generally assumed to fluctuate very little.

lined with concrete, the soils are generally very high in phosphorus as a result of continuous manure enrichment; where they are lined, their imperviousness leads to direct runoff. Milkhouse waste and silage leachate have also been identified as major sources of farmstead phosphorus loss in the Lake Champlain Basin (Gaddis and Voinov 2010). However, these sources are generally only significant during storm events and behave much more like point sources; while the event discharges can be very large, their contribution over the course of the whole year is generally small. Manure storage systems are also a concern, but they are required for farms falling under the jurisdiction of the Environmental Protection Agency's (EPA) National Pollutant Discharge Elimination System (NPDES) permit program, and therefore are highly engineered, and have low failure rates. Manure storage systems on farms that are not regulated (fewer than 200 milking cows) are of more concern, because the laws governing their design and use are either non-existent or not particularly stringent.

Though agriculture is typically thought of as the major contributor of phosphorus to Lake Champlain, the truth is somewhat more complex. While the proportion of agricultural land in most watersheds exceeds that in the urban category, urban lands contribute several times the amount of phosphorus per unit land area, and can therefore contribute equal or greater total annual loads of phosphorus (Paul and Meyer 2001). Therefore, on a basin-wide scale, the distribution of loads between these two sources is more equal than land use totals alone would predict. Agricultural and urban land uses are now estimated to contribute 33% and 18% of total loads, respectively (approximately 43% and 24% of land-based phosphorus exports), though within watersheds the balance may shift heavily toward one land use or the other (Tetra Tech 2013).

Urban Sources: Impervious Surfaces

As urban development has spread around the country, resource managers and academic scientists alike have become aware of major changes that occur in stream ecosystems as a direct result of the loss of the soil's ability to absorb precipitation. This loss generally comes as a result of the development of impervious surfaces (roads, sidewalks, parking lots, building footprints, etc.), and the severe compaction of adjacent unpaved soils to a point of near-imperviousness. Streams in urbanized catchments undergo profound changes as the extent of imperviousness increases: the cross-sectional morphology becomes more incised, setting off major bouts of bank erosion; water and sediment delivery to the stream become more punctuated, with steep hydrograph peaks and falls during storm events; temperatures become more extreme, which in turn can change nutrient processing rates; and delivery of toxic contaminants and organic compounds increases, lowering biotic richness (Paul and Meyer 2001, Meyer et al. 2005, Walsh et al. 2005).

Even for unpaved areas in urban landscapes (lawns, urban open space, greenways, etc.), close physical proximity to impervious surfaces subjects them to many of the same losses in the soil's ability to drain and store water. During storm events, these soils reach saturation much more quickly, and contribute to runoff at rates much higher than undisturbed soils (Booth and Jackson 1997). In the Lake Champlain Basin, this is especially problematic because much of the land currently under development was previously agricultural, and the soils are likely to have high phosphorus concentrations. Additionally, lawn and landscaping preparation for new houses (as well as established houses) use fertilizer at rates matching or exceeding agricultural rates for row crops, often with far less education and information (Barth 1995). Ditching, tiling, curbs, and storm drains installed as part of most residential and commercial development encourages rapid transfer of stormwater offsite to a stream system, or at best, a stormwater collection system. The often high inputs of phosphorus to lawns, combined with the already high native phosphorus levels of soils in the Champlain Basin, act as a significant source of both dissolved and particulate phosphorus.

Particulates common on road surfaces such as leaves (during the fall) and road sand (during the winter and spring) can also present significant sources of phosphorus. Gaddis and Voinov (2010) measured the phosphorus concentration of road sand applied in St. Albans, Vermont to be 0.78g/kg, or 1.55 lbs P/ton of sand. The town of St. Albans alone applies 600 tons of road sand per year, which totals nearly a half ton of phosphorus. The total for the basin as a whole could be an appreciable percentage of the total phosphorus load.

Urban Sources: Infrastructure

Many older cities across the U.S. installed combined sewer systems as a more economical alternative to building separate systems for wastewater and stormwater before it was recognized that overflows from these combined systems presented a high degree of risk to quality of their receiving waters (Field and Struzeski 1972). Municipalities around the Champlain Basin are no exception, and despite the efforts of state and municipal governments to separate stormwater and wastewater systems, combined sewer overflows still discharge untreated wastewater during high flow events. Discharges from these combined systems not only occur at the pipe outfalls, but also where the aging sewer pipes leak or break underground (Phillips and Chalmers 2009); these places are typically only detectable by thorough field sampling and analysis, such as by an Illicit Discharge Detection and Elimination program (VTANR 2010).

Stream Bank Erosion & Channel Migration

Though streams have generally been considered conduits for the transport of sediment and phosphorus from upland sources to the lake, stream channels themselves could be contributing significantly to phosphorus loading. In 2004 Vermont's River Management Program began conducting geomorphic assessments of a large number of streams to characterize the stability of Vermont's stream network. Over the course of the next several years, nearly 1400 miles of river were assessed, and the data suggest that 75% of stream miles in Vermont are entrenched and unstable, either vertically, horizontally, or both. Using a combined modeling and field data approach to characterize the contribution of streambank erosion to suspended sediment and phosphorus loads at the mouth of the Missisquoi River, Langendoen et al. (2012) found that as much as half of the total phosphorus load for that tributary may be derived from bank erosion and channel migration.

The large variability between watersheds in the Lake Champlain Basin in terms of their physical geography and land uses, and the rapid development of these characteristics over time has created a diffuse phosphorus loading problem of a varied nature; not only is the magnitude of the phosphorus problem in each tributary different, the nature and potential solutions are variable as well. In short, the phosphorus problem in Lake Champlain is complex, and any solution that is to be effective must be complex as well.

Managing Phosphorus: Research, Regulation, & Management

Identifying the Problem & Creating a Governance Structure

Symptoms of excessive phosphorus loading to Lake Champlain first became evident in the 1970's, and efforts to understand the problem in a comprehensive way also began then. A series of studies to estimate the amount of phosphorus entering the lake were completed over the middle of the decade, culminating in the Lake Champlain Basin Study

(1979). This study compiled data from many sources, and provided a basic plan for stabilizing or reducing mainly point source phosphorus inputs through at least 1990. It provided evidence to support the regulatory programs in place already, such as Vermont's Act 250, and the ban on phosphorus based detergents in Vermont and New York, and additionally suggested building phosphorus removal capability into several of the wastewater treatment facilities and initiating agricultural non-point source control incentive programs (Lake Champlain Study 1979). Many of these recommendations had been essentially completed by the mid 1990's, though the eutrophication symptoms had not improved significantly to that point (Smeltzer 1997).

In 1988, the states of Vermont, New York, and the province of Quebec signed a Memorandum of Understanding (MOU) that initiated a process to establish in-lake criteria for phosphorus concentrations, and establish target watershed loadings to achieve the in-lake criteria (Stickney et al. 2001). The Lake Champlain Steering Committee was established at this point to guide that process. The Lake Champlain Special Designation Act of 1990 created a group responsible for developing a comprehensive plan to prevent and control phosphorus pollution that would lead to restoration of the lake (the Lake Champlain Management Conference), and to coordinate that effort and all other efforts between the three jurisdictions for research, demonstration projects, lake and tributary monitoring, and education and outreach initiatives. The Lake Champlain Basin Program was established to staff the work of the Management Conference and it continues to act as the coordinating body between Vermont, New York, and Quebec for efforts to manage the lake's natural and cultural resources.

Establishment of a Research and Monitoring Program

Through the end of the 1990's, a flurry of studies on Lake Champlain's phosphorus problem were conducted and published, but they primarily focused on 1) identifying research questions, 2) identifying and quantifying phosphorus sources, or 3) understanding phosphorus dynamics both in streams and in the lake. With few exceptions, little scientific work at the time focused on ecosystem responses to management actions (Table 1-1). During this period, management actions centered around regulated phosphorus sources such wastewater treatment, phosphorus-containing detergents, and structural practices (e.g., manure pits) for regulated farms (Smeltzer 1997, Lake Champlain Basin Program 2000). Over the next decade, the research focus in the basin shifted only slightly. Efforts to document the extent of key sources and to understand the behavior of phosphorus on the land, in streams, and in the lake became more refined (McDowell et al. 2002, Troy et al. 2007, Phillips and Chalmers 2009). Other research topics began to emerge during this time period: proposals for integrated assessment criteria (Watzin et al. 2005), evaluation of new policy tools and legislative options for controlling phosphorus (Winsten 2004), and some studies of the effects of management actions on agricultural land uses in particular (Meals 2001, Contach et al. 2003, Michaud et al. 2007, Bushey et al. 2009), were all published during this time.

Table 1-1. Technical reports and scientific research on Lake Champlain phosphorus loading, 1992-1999.

Source	Title	Focus
LCBP Technical Report No. 1, 1992	A Research and Monitoring Agenda for Lake Champlain. Proceedings of a Workshop, December 17-19, 1991, Burlington, VT. Lake Champlain Research Consortium.	Planning, Monitoring
LCBP Technical Report No. 2, 1993	Design and Initial Implementation of a Comprehensive Agricultural Monitoring and Evaluation Network (CAMEN) for the Lake Champlain Basin.	Planning, Monitoring
LCBP Technical Report No. 6, 1994	Lake Champlain Nonpoint Source Pollution Assessment.	Identification and Quantification of Sources
LCBP Technical Report	(a) Dynamic Mass Balance Model of Internal Phosphorus	Identification and

No. 7a, b, c, 1994	Loading in St. Albans Bay, Lake Champlain. (b) History of Phosphorus Loading to St. Albans Bay, 1850-1990. (c) Assessment of Sediment Phosphorus Distribution and Long-Term Recycling in St. Albans Bay, Lake Champlain.	Quantification of Sources
LCBP Technical Report No. 19, 1996	Hydrodynamic and Water Quality Modeling of Lake Champlain.	Identification and Quantification of Sources
LCBP Technical Report No. 20, 1996	Understanding Phosphorus Cycling, Transport and Storage in Stream Ecosystems as a Basis for Phosphorus Management.	Understanding Phosphorus Dynamics
Meals, D. W. 1996. Water Science and Technology **33**:197-204.	Watershed-scale response to agricultural diffuse pollution control programs in Vermont, USA	Management Effects
Weller, C. M., M. C. Watzin, and D. Wang. 1996. Environmental Management 20:731-739.	Role of wetlands in reducing phosphorus loading to surface water in eight watersheds in the Lake Champlain Basin.	Understanding Phosphorus Dynamics
LCBP Technical Report No. 22, 1997	Characterization of On-Farm Phosphorus Budgets and Management in the Lake Champlain Basin.	Identification and Quantification of Sources
LCBP Technical Report No. 25	Urban Nonpoint Pollution Source Assessment of the Greater Burlington Area: Urban Stormwater Characterization Project.	Identification and Quantification of Sources
Levine, S. N., A. D. Shambaugh, S. E. Pomeroy, and M. Braner. 1997. Journal of Great Lakes Research 23:131-148.	Phosphorus, nitrogen, and silica as controls on phytoplankton biomass and species composition in Lake Champlain (USA-Canada).	Understanding Phosphorus Dynamics
LCBP Technical Report No. 29, 1997	Evaluation of Soil Factors Controlling Phosphorus Concentration in Runoff from Agricultural Soils in the Lake Champlain Basin. Frederick R. Magdoff, William E. Jokela and Robert P. Durieux, University of Vermont, Department of Plant and Soil Sciences. June 1997.	Understanding Phosphorus Dynamics
LCBP Technical Report No. 26, 1998	Long-Term Water Quality and Biological Monitoring Project for Lake Champlain. Cumulative Report for Project Years 1992-1996.	Planning, Monitoring
Meals, D. W. and L. F. Budd. 1998. Journal of the American Water Resources Association 34:251-265.	Lake Champlain Basin nonpoint source phosphorus assessment.	Identification and Quantification of Sources
LCBP Technical Report No. 31, 1999	Estimation of Lake Champlain Basinwide Nonpoint Source Phosphorus Export.	Identification and Quantification of Sources
LCBP Technical Report No. 35, 1999	Determination and Quantification of Factors Controlling Pollutant Delivery from Agricultural Land to Streams in the Lake Champlain Basin.	Understanding Phosphorus Dynamics
LCBP Technical Report No. 36, 1999	Cost-Effective Phosphorus Removal from Secondary Wastewater Effluent through Mineral Adsorption.	Management Effects

A Shift in Management Priorities

The collective focus of management agencies over the next decade shifted strongly toward treatment of nonpoint sources of phosphorus, once it was realized that the fraction of the total phosphorus load attributable to wastewater treatment was dropping below 5% (VTANR & VTAAFM 2011). Regulatory changes to the EPA's NPDES program to include high density animal farms (CAFOs) in 2003 also gave the states of New York and Vermont regulatory authority to require a suite of structural BMP's to manage manure storage and barnyard runoff, and to implement and maintain soil and nutrient conservation practices such as Nutrient Management Plans (NMPs) and riparian buffers.

Though these changes affected the larger farms in Vermont and New York, many farms still remain relatively small; at latest count, there were 175 regulated farms in Vermont [Large and Medium Farm Operations (L/MFOs)], and an estimated 900 unregulated Small Farm Operations (SFOs). Though all farms are required to abide by a set of minimal Accepted Agricultural Practices (AAPs), the vast majority of field- and farmstead-based practices shown to be the most effective at controlling edge-of-field sediment and phosphorus losses are voluntary under the current regulatory structure; adoption of these practices has understandably been slow and sporadic. Additionally, and perhaps more importantly, there was little effort to document the effects of these practices at the watershed scale with the exception of two studies by Meals (1996, 2001). However, each of these studies implemented a suite of practices simultaneously, and was therefore not able to distinguish relative effectiveness between individual or combinations of practices.

There are two major drawbacks to the management strategy collectively employed by local, state, and federal agencies over the past 20 years. Firstly, the reliance on dealing with the "low-hanging fruit" – the existing point source regulatory solutions – despite the recognition that non-point sources were more extensive and would take longer to address has delayed the attainment of in-lake phosphorus criteria. Secondly, there is no defined structure to enable estimation of the effects of management actions in an effort to either find the most efficient solutions or to provide the best performance across other important objectives. Though studies initiated by the LCBP in the past few years have aimed to take a stronger watershed-scale management focus, there is still no formal plan for using this data to inform management, or to allow extrapolation to larger watersheds or the basin as a whole. There is little effort to include simulation, experimentation, or active learning as part of the decision-making process. Management agencies in each jurisdiction are aware of monitoring data and analyses (Smeltzer and Simoneau 2008, Smeltzer et al. 2009, Medalie et al. 2012, Smeltzer et al. 2012), but the connection to management action is weak.

In recognition of this, the Lake Champlain Steering Committee charged the LCBP in late 2009 with developing an adaptive management framework for the control of phosphorus loading. The principles of adaptive management hold potential to improve the management of phosphorus in the Lake Champlain Basin in two major ways: first, adaptive management is a process that produces transparent and defensible management decisions through the development of clear management goals and careful analysis of alternative courses of action including any relevant uncertainties regarding the outcomes; second, these key uncertainties, for example about the effects of management, are reduced over time in a planned and deliberate fashion. These features hold great appeal for management agencies in the Lake Champlain Basin given the ongoing revision of Vermont's TMDL, and the public awareness of the relatively slow progress in improving water quality. I provide below a review of the principles of structured decision-making and adaptive management and their application to managing phosphorus in the Lake Champlain Basin.

Adaptive Management: Confronting the Realities of Managing Large Ecosystems

Adaptive Management has its roots in the practice of Adaptive Environmental Assessment and Management, which was developed in response to calls from the resource management community for a method to aid decision-making about management actions, given the range of uncertainties inherent in managing large and complex ecosystems (Holling 1978). In its modern form, adaptive management is defined as a highly structured, well-planned cycle of "learning by doing" that combines the experimental and analytical elements of the scientific method with the carefully articulated structure of project management and decision analysis. While there is wide opportunity for interpretation in how many steps there should be, what they are called, and how they are divided or clumped, the scientific and professional literature widely agrees that adaptive management must consist of iterations between a transparent decision making component and an opportunity for learning through inference (figure 1, from Allen et al. 2011). In addition, adaptive management can often include a double-loop construction, such that the technical learning aspects occur several times before the objectives are revisited (Williams 2011a). The exact way in which the steps are laid out and the relative emphases of each part of the process determine in part the rate at which learning can occur.

For the tools of adaptive management to prove helpful, the foundations of two key processes must be present. The first is a process for making decisions that considers multiple value-based objectives at the same time, evaluates action alternatives relative to each other, and that produces repeatable and defensible results. The tools that enable this sort of process are referred to in general as Structured Decision-Making (SDM) methods (Gregory et al. 2012). SDM methods have been successfully used in a wide variety of situations, and increasingly are being used by the U.S. federal government in natural resources management as a way to make more informed and defensible decisions about the best use of public resources (Stankey et al. 2005, Williams and Brown 2012), including use by the EPA in coastal watershed management applications (Carriger et al. 2013).

In the context of Lake Champlain, the use of SDM methods requires an understanding of what management goals are most appropriate from both a scientific and a public perspective – i.e., what do the data suggest are appropriate goals, and what goals do the public and other stakeholders want to see achieved. A large body of research has shown that explicitly including these sorts of values into decision-making improves the quality of the outcomes (Keeney 1992). However, to do that, decision-makers need to be clear about what value-based goals are important at the outset of the decision-making process. This information paves the way toward better generation of management options and better ability to make informed tradeoffs later in the decision process.

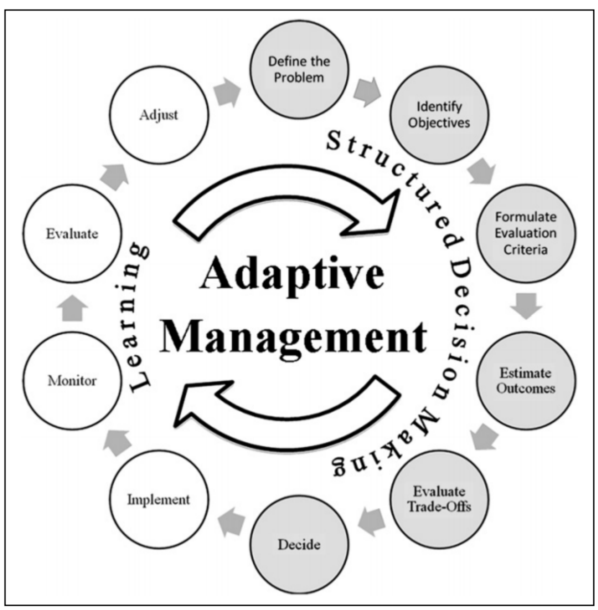

Figure 1-2. Adaptive management cycle showing iteration between phases relying on structured decision-making and learning through inference (Allen et al. 2011)

The second key process for successful application of adaptive management tools is a way to learn about management actions through inference; generally this is accomplished by using statistical methods to compare the predicted outcomes of management actions to their actual monitored outcomes (Walters 1986). Resource managers often rely on complex computer models to generate these predictions, but research and experience have shown that relatively simple estimations that use basic methods and existing data can often provide enough discrimination among alternatives to enable more informed decisions, with lower investments of time and money.

The method has been applied widely to problems where 1) a high degree of uncertainty about the ecological system exists, 2) where there are many stakeholders and multiple (and potentially competing) objectives for the relevant management agencies and 3) those agencies make recurring management decisions about the same resource. Common past uses have included managing harvest limits for ungulates, migratory birds, and fisheries (particularly salmonids), managing large timber tracts, and managing water use in arid climates (Johnson and Williams 1999, Walters 2007, Wichelns 2010, Williams and Brown 2012). Adaptive management has been used as a tool in each of these situations to maximize both economic and environmental sustainability. In general, these adaptive management approaches have met with good success where the spatial scale is small and the controllability of the system and management activities is high, and with moderate success where the system is very large, or the controllability is very low.

The explicit structure of adaptive management provides learning opportunities at each pass through the cycle. If they are taken, these opportunities can reduce the usually high degree of uncertainty in managers' and scientists' understanding of the effects of their management actions, and of the system itself. A good understanding of the source of important uncertainties is essential for successfully reducing their magnitude. Regan et al. (2002) identify several dimensions of uncertainty that can be applied to water quality management in the Lake Champlain basin. Adaptive management and structured decision making can help to identify and potentially reduce these uncertainties (Table 1-2).

Table 1-2. Categorical sources of uncertainty addressed through adaptive management, adapted from Regan et al. (2002).

Source of Uncertainty:	Definition	Methods for Expression
Natural variation and randomness	Derives from unpredictable changes in ecosystem states due either to complex changes in precursor conditions, or to a lack of deterministic dependency.	Probability distributions, confidence intervals
Measurement error and bias	Derives from imperfect estimation of the values associated with ecosystem states – can be random (error) or systematic (bias). Expressed as the statistical variance.	Confidence intervals (for error) and detection and removal methods (for bias)
Model and structural uncertainty	Derives from our incomplete representation of environmental and ecological relationships	Validation studies, alternative model forms, clear statements of assumptions
Linguistic uncertainty and context dependence	Derives from inadequate specification of conditions important for proper understanding and use of information	Full description of bounds of information applicability

Despite a solid and consistent theoretical foundation, various applications of the adaptive management concepts often emphasize certain elements of the process over others. While this flexibility is important to ensure that the approach is adaptable to new problems, these different emphases have implications for the rate at which learning occurs and the defensibility of the decision-making process.

The degree to which the learning portion is emphasized through the creation of models, testing of a set of alternative hypotheses, and reducing uncertainty is a primary element that receives different levels of attention across applications (Williams 2011b). The most common method of management (i.e., not adaptive management) is one where hypotheses are not clearly stated (either in a classical experimental design setting or in models), and management options are evaluated one at a time (Figure 1-3). In this method, management options are only rejected upon their failure, which could take months, years, or decades to discover, depending on the resource and the monitoring program (Meals et al. 2010), and therefore learning is exceedingly slow, if it occurs at all. It is also a risk-prone method of management, despite the popular perception that it is risk-averse; a management failure within this method can be both financially costly (because of effort wasted) and environmentally devastating (from time wasted and lack of effective management).

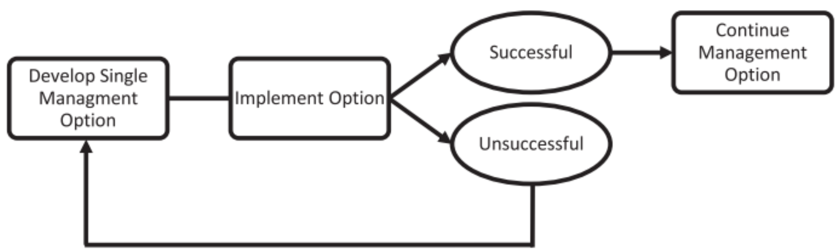

Figure 1-3. Trial-and-Error Learning, where a single management option is implemented, and rejected if unsuccessful options are rejected in favor of a new management options (Allen et al. 2011).

The development of multiple alternative management choices at the outset of the project choices at the very least saves time when management failures occur, as in Figure 1-4, though learning is still slow. While passive adaptive management does recognize that management affects resources (and plans for that by developing multiple management options), it is not able to reduce uncertainty in a systematic manner (Williams 2011b). In addition, opportunities for statistical inference are rare in this mode because of the lack of defined treatments and the lack of replication. There are situations where passive adaptive management is warranted. When governance structures or ecosystem scale prohibit explicit experimentation, passive adaptive management can be used to anticipate and account for changes in resources as a result of management action. However, passive adaptive management most often occurs when experimentation and monitoring (i.e., active adaptive management) is not supported.

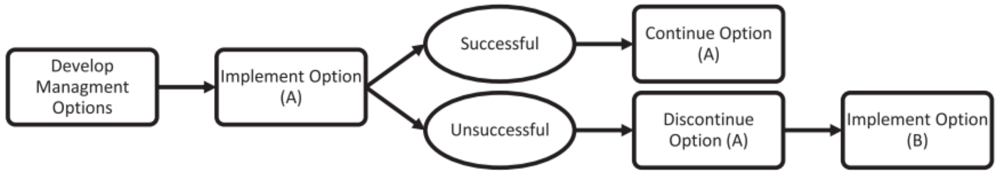

Figure 1-4. Passive Adaptive Management, where multiple management options are developed, and one is chosen for evaluation at each time step; unsuccessful options are rejected in favor of a new, previously developed alternative (Allen et al. 2011).

Active adaptive management (Figure 1-5), which is the most complex form both scientifically and organizationally, has the highest potential to yield results quickly, because it consistently tests larger numbers of competing scientific hypotheses in experimental or quasi-experimental settings (Gregory et al. 2006). Management actions and policies are treated as hypotheses; a particular action is taken in a particular location for a particular reason, and this action serves as a treatment. Implementation of that treatment across the landscape can serve as replication (depending on the experimental design). In this context, management treatments are compared and tested against each other rather than against a control (Blumstein 2007). Alternatively, these hypotheses can take the more traditional form of clearly stated expectations of management effects or a series of competing predictive models, such as those used by Martin et al (2011) to predict the effects of hiking restrictions on golden eagle nest occupancy and breeding success rates (McCarthy and Possingham 2007). The use of statistical inference (often through advanced methods) is a major part of the evaluation phase in this mode. One major advantage of this method is that communication about the basis of management decisions is very clear, and fully defensible (e.g., U.S. Fish and Wildlife Service 2011). This is advantageous because it makes transparent many of the value-based elements of decision making, which if left implicit can be perceived as faulty decision-making by stakeholders who hold different values. Additionally, this process reveals the relative importance of reducing ecological uncertainty or adjusting management objectives and constraints in achieving the desired management outcomes (Nichols et al. 2007).

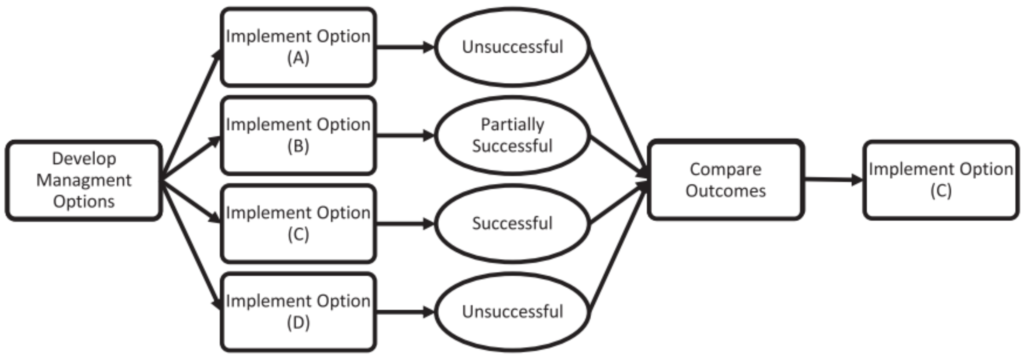

Figure 1-5. Active Adaptive Management, where multiple (simultaneous) management actions are evaluated and compared against each other, and the most effective one chosen for continuation (Allen et al. 2011).

Because adaptive management is explicitly not a "one-size-fits-all" approach, the ways in which adaptive management has been implemented represent a wide spectrum, and clear categorization is difficult. There are some lessons to be learned from the successes and failures of some implemented adaptive management frameworks, and themes have emerged from these cases that can help inform the development of new frameworks and the improvement of existing ones.

Theory and Reality: The Successes and Failures of Implemented Adaptive Management

Though the conceptual underpinnings of adaptive management are strong and there is a general consensus about the required elements of its application, there are surprisingly few good examples of the practice of adaptive management at large scales, particularly in water resource management (Walters 1997). Successful applications have primarily been in specific situations where the resource in question responds well to management actions (i.e., controllability is high), while unsuccessful applications occur in situations where the resource in questions responds very slowly to management (or not at all).

It is becoming increasingly clear that while adaptive management holds great potential, it should not be applied to all situations (Figure 1-6). Where uncertainty is low, there is no need for adaptive management. Likewise, where management is not expected to have any observable effect on the system, little learning is possible. Other tools, such as scenario planning or hedging (Peterson et al. 2003), are easier to implement and better-equipped to deal with the particulars of low control systems. The level of risk can also be a factor in the decision whether to use adaptive management. Where there is a high risk to the environment of failing to reach the management objectives, adaptive management can increase the likelihood that objectives are met over the long term.

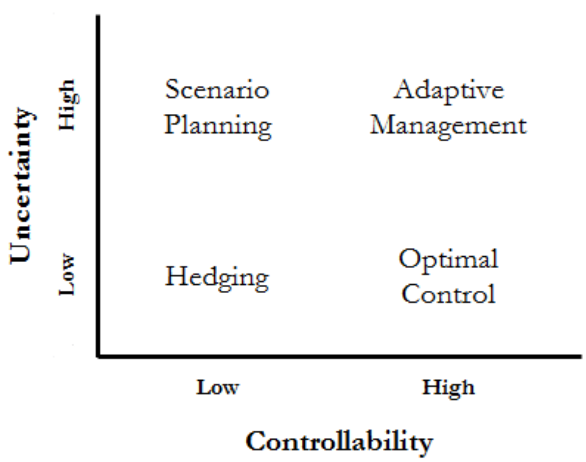

Figure 1-6. Role of uncertainty and controllability in determining whether adaptive management is a suitable approach (adapted from Peterson et al. 2003).

Several authors have suggested reasons for the failure of adaptive management plans (Gunderson et al. 1995, Walters 1997, Schreiber et al. 2004, Gregory et al. 2006, Allen and Gunderson 2011), and common themes emerge from these critiques (Table 1-3). While some failings are due to poor rigor or inappropriately applied plans (technical failures), most are related to problems not addressed in the planning phase (planning failures), problems that arise from misunderstandings and poor communication (communication failures), or problems that arise from failures of the governance structure (governance failures).

Table 1-3. Common reasons for failure of implemented adaptive management plans (Walters 1997, Schreiber et al. 2004, Gregory et al. 2006, Eberhard et al. 2009, Allen and Gunderson 2011)

Category	Source of Failure
Technical Failures	• The system is too large; system responses from management are too slow to be detected • Tested management experiments do not produce a large enough effects to detect change
Planning Failures	• Lack of confidence in assessments due to lack of technical rigor (e.g., in uncertainty analyses, modeling) • Sources of uncertainty and constraints were not accounted for, including social constraints • Sources of uncertainty not targeted for reduction
Communication Failures	• Unclear analysis of timeframe, cost, benefits • Poor articulation of objectives • High perceived risk of failure • Lack of stakeholder involvement • Adaptive Management seen as a threat to classic hypothesis-driven experimental science • Agencies/Organizations unwilling to admit uncertainty
Governance Failures	• Initial prescriptions are followed rather than updating management strategies according to what is learned and the AM protocol • Action is put off until "better science" is available • Focus on planning, not action • Politically sensitive findings are suppressed or compromised to a point of ineffectiveness • Institutional "memory loss", especially for long-term frameworks • Self-interest in partner organizations does not encourage solving the problem

In large ecosystems (e.g., the Columbia River Basin, the Everglades, Chesapeake Bay) the vast scale of ecosystem processes produce long time lags between management actions and system response. In these situations, the design of the monitoring program must be matched to the spatial and temporal scale of the ecosystem changes. The lack of ability to detect environmental change through monitoring may be caused by any combination of a poor monitoring program, inappropriate statistical questions, and large time lags between management action and environmental change (Marcot 1998, Gregory et al. 2007, Meals et al. 2010). When monitoring programs are well designed and management questions are aimed at appropriate scales, the adaptive management process can be very successful even in large ecosystems. For example, Glen Canyon Dam Adaptive Management Program is commonly cited as one of the best examples of active adaptive management currently being practiced in the US (Pulwarty and Melis 2001, Owens 2009).

While there are many examples of failed adaptive management programs and many reasons for these failures, many of the problems discussed above are avoidable. Greater attention to detail in the planning phase, a stronger emphasis on clear communication, and good design of monitoring programs can all help to provide a structure to deal with ecological or governance surprises found as the framework plays out.

While these threats to success are real, the Lake Champlain Basin remains a candidate for the use of adaptive management concepts to manage phosphorus loadings. The existing governance structure in the Lake Champlain basin, represented in part by the LCBP, is small and flexible enough to accommodate an SDM approach to decision-making, and the wealth of science in the basin is of great value.

Firstly, though the watershed is large, the long history of scientific work to characterize the watershed and its processes has produced a wealth of information. Perhaps most significantly, the Lake Champlain Basin Program has supported a Long-Term Monitoring Program since its inception twenty years ago. This monitoring program has characterized total and dissolved phosphorus from each major tributary for that period. Additionally, a database containing more than 60 technical reports published by the LCBP (Lake Champlain Basin Program 2012), and the large volume of academic work done on the lake have uncovered important ecological information about food web and trophic interactions (Mihuc et al. 2012), phosphorus mobility in stream channels (Langendoen et al. 2012), the extent and locations of key phosphorus source areas in selected watersheds (Ghebremichael et al. 2010, Winchell et al. 2011).

This information has given us valuable insight into the ecological connections between the watershed and the lake, but it has also given insight into which categories of uncertainty may be harder to reduce in a short time frame. For example, the water quality effects of watershed scale implementation of suites of agricultural BMPs have proven difficult to assess here, and the reasons are also hard to assess (Meals 1996, 2001, Meals et al. 2010). Avoiding disproportionately high expenditures on reducing uncertainties that will yield little to no information are an avoidable pitfall here. A careful analysis during the planning phase of key uncertainties that are likely to be reducible will save time and money in the long run.

Secondly, the Lake Champlain Basin has a strong history of communication with stakeholders and the public. The structure and membership of the various committees that the LCBP supports (Figure 1-7), such as the Steering Committee and the Citizens Advisory Committees show a commitment to stakeholder inclusion. Additionally, the commitment to informing the public is strong, shown by the regularity of publications intended for the public that are transparent in their presentation of goals, objectives, and methods for achieving those goals (Lake Champlain Basin Program 1996, 2003, 2010) and in presenting the results of monitoring and evaluations of the health of Lake Champlain (LCBP 2008, 2012). This focus on inclusion and communication has existed since the inception of the LCBP, and is unlikely to change.

Additionally, many of the governance issues presented above are alleviated to some degree by the structure of the LCBP and the agreements between the jurisdictions. The agreements between Vermont, New York, and Quebec have been and remain non-binding and voluntary (Stickney 2008), which has produced a culture of lake management and stewardship that relies on the trust and consensus of the partner entities. While non-binding and voluntary agreements might have the potential to de-emphasize the accomplishment of difficult objectives, the high public profile nature of the Lake Champlain eutrophication problem demands action by those in management positions.

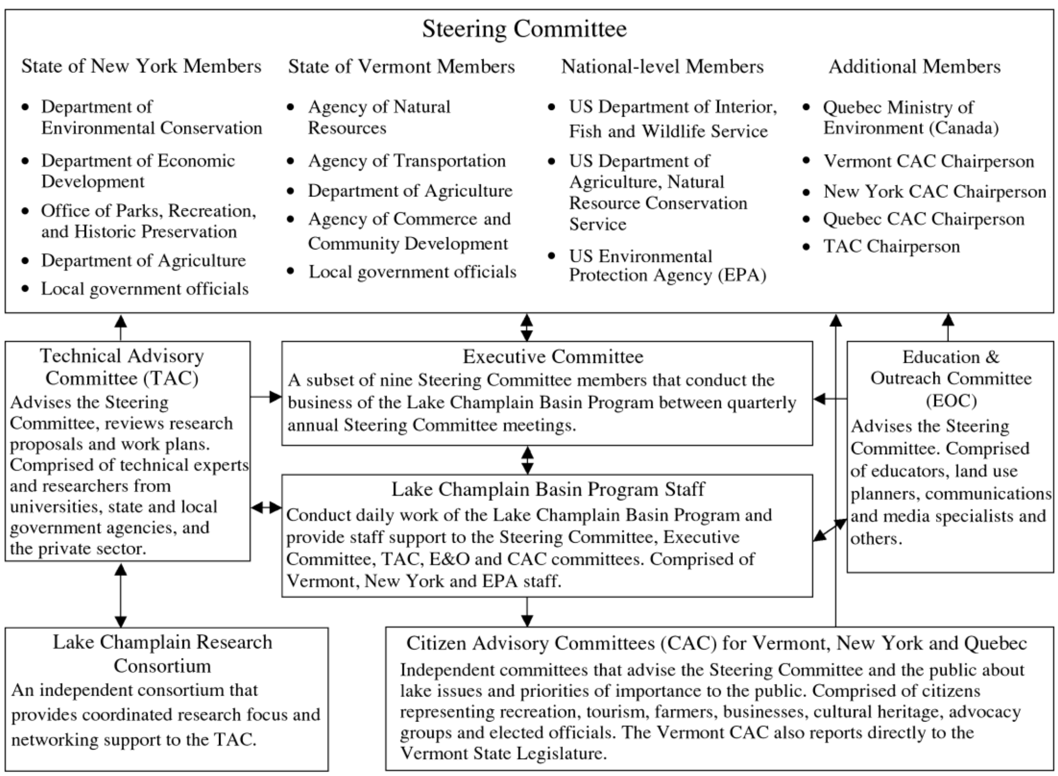

Figure 1-7. The committees supported by the Lake Champlain Basin Program are collectively a forum for the diverse stakeholder interests of the Basin as a whole (Stickney et al. 2001)

The Lake Champlain Basin has many of the components necessary for the successful application of tools for the adaptive management of phosphorus; the major gap to date has been a lack of transparent and rigorous decision-support tools that can integrate all of the relevant monitoring data (both implementation and ecological data), glean information from that data, and then use that information to make good predictions of the effects of management decisions on relevant management objectives.

The Role of Decision Support Tools

The basis for decision-support tools lies in four decades of research in the field of behavioral psychology that shows convincingly that when faced with complex situations, people are not good at making rational decisions without help (Slovic et al. 1977). In particular, two features of our cognitive processes hinder rational decision making. First, when faced with complex contextual information, most people resort to simple, low-effort heuristics to guide their decision making and judgment. Through a series of experiments Tversky and Kahneman (1974) showed that the way people process uncertain information is sensitive to past experiences with similar information, and their ability to imagine the context. While these heuristics do provide good decisions in some cases (Gigerenzer 2014), they are implicit by definition, and this lack of transparency limits their acceptability for guiding public policy decisions (Slovic et al. 1977).

Second, other research has shown that in addition to being poor intuitive statisticians, most people have little sense of their preferences for the outcomes of their decisions until they are asked for them, and that preferences are only solidified on an as-needed basis in response to the decision context (Slovic 1995). Because they are amorphous at any particular time, our preferences are also fluid even over short periods of time, which means that without explicit statements of what they are, the implicit criteria for making decisions changes over time. Our poor intuitive understanding of our preferences and little intuitive understanding of probabilities means that our perceptions of risk bear little resemblance to analytical calculations of risk, with sometimes disastrous consequences to the environment and to ourselves (Slovic 1987).

These two major shortcomings in our ability to make rational decisions on our own have clear implications for management decision making in the Lake Champlain basin. Estimates of phosphorus loads, the effectiveness of various management practices, and the extent to which we can control phosphorus movements through the watershed are all subject to various sources and magnitudes of uncertainty, and relying on our intuition or on simplistic tools that ignore that uncertainty almost certainly means systematic errors in judgment about the best course of action (Morgan and Henrion 1990). Likewise, the cost and effectiveness of management practices are two important considerations in the choice of a management strategy, but implicit assumptions about their value relative to each other or relative to different realized levels of cost and effectiveness mean that a management strategy that provides the best compromise between those two objectives is hard to identify (Keeney 1992).

In an effort to help formalize and make explicit many of the internal judgments that are involved in making decisions, a wide variety of analytical tools have been developed to aid decision makers in understanding their own values in the context of setting objectives and making informed trade-offs between objectives, and in comprehending and accounting for uncertainty in predicting the outcomes of decisions (von Winterfeldt and Edwards 1986, Keeney 1992, Kirkwood 1992b, Goodwin and Wright 2004). Decision support methods for making rational trade-offs and for dealing with uncertainty are discussed briefly below.

There is a wide range in complexity of tools for making informed and rational tradeoffs. At one end of the spectrum is the intuitive and analytically simple bartering process referred to as "even swaps", where the decision-maker is forced to think about the relative values of objectives and trade between them (Hammond et al. 1998). In the middle of the spectrum is multiple-criteria decision analysis, which uses a system of normalizing the predicted consequences across alternatives, and then weighting objectives to provide an overall utility for each alternative across objectives (Goodwin and Wright 2004). At the farthest end of the complexity spectrum is optimization, which is a computationally intensive process of finding optimal combinations of decisions and constraints when the relationships between objectives can be non-linear.

There is a wide range in complexity and analytical rigor among decision aids that demonstrate and incorporate the effects of uncertainty into decision making. The simplest of these involve approximations of the possible ranges of uncertain quantities and evaluation of the impact of points along that range (e.g., Kirkwood 1992a). However, the integration of the Bayesian statistical paradigm into the decision analysis framework has yielded a variety of improvements in the ability of decision analytic techniques to accommodate uncertainty (Varis 1997). In particular, Bayesian networks have gained wide use over the past decade in natural resource management situations because of their rigorous treatment of uncertainty their flexible nature (McCann et al. 2006, Jensen and Nielsen 2007). In particular, Bayesian networks are able to represent decision problems through intuitive causal or correlational relationships that link decision points with relevant outcomes. Because these networks are able to propagate uncertainty through the elements of the network, they present decision makers with a range of possible outcomes resulting from any given choice. These features have direct value for decision-makers predicting the outcomes of environmental management decisions because of the large role that uncertainty plays in those kinds of decisions.

Trajectory of this Dissertation

In light of the issues described in this review, my goal here is to demonstrate an approach to filling the gap between the wealth of watershed data that often exists and the kinds of decisions that water resource managers actually face in managing water quality. Though this dissertation focuses on Lake Champlain as a test case, this gap exists in many other similar situations around the country.

Chapter 2 demonstrates a simple spreadsheet-based method for identifying the areas of greatest potential for further phosphorus reductions, for estimating the potential scale of those reductions, and for identifying the significant tradeoffs that exist between cost and effectiveness at high levels of management.

In Chapter 3, I develop and apply a Bayesian hierarchical modeling approach to estimate land-use specific phosphorus loading rates in an effort to quantify the variability of NPS contributions over time and space. This approach incorporates annual variability and uncertainty about land use areas into estimates of the relative contributions to total phosphorus loads of different land uses and watershed-scale residual loading.

In Chapter 4, I develop and apply a Bayes network to predict the effects of alternative management scenarios on phosphorus loads under conditions of uncertainty in land use based loading rates, in the effects of management, and in the cost of management. Using evolutionary optimization and multiple-criteria decision analysis, I explore the tradeoffs between cost, effectiveness, and distributional equity in the burden of management.

Chapter 5 summarizes the work in the context of the current state of watershed management for water quality. I discuss the existing uncertainties and shortcomings in the scientific basis for the current use of the Command and Control approach to enhancing water quality, and then identify several alternative approaches to improving decision-making for water quality restoration.

References

Allen, C. R., J. J. Fontaine, K. L. Pope, and A. S. Garmestani. 2011. Adaptive management for a turbulent future. Journal of Environmental Management **92**:1339-1345.

Allen, C. R., and L. H. Gunderson. 2011. Pathology and failure in the design and implementation of adaptive management. Journal of Environmental Management **92**:1379-1384.

Anderson, B. H., and F. R. Magdoff. 2000. Dairy farm characteristics and managed flows of phosphorus. American Journal of Alternative Agriculture **15**:19-25.

Barth, C. A. 1995. Nutrient Movement from the Lawn to the Stream. Pages 239-249 Watershed Protection Techniques. Center for Watershed Protection.

Blumstein, D. T. 2007. Darwinian decision making: Putting the adaptive into adaptive management. Conservation Biology **21**:552-553.

Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems: Degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association **33**:1077-1090.

Bushey, J. K., J. E. Carter, J. R. Chamberlain, and S. A. Flis. 2009. Reducting Phosphorus Runoff from Small Livestock Farms into Missisquoi Bay. Technical Report No. 58. Lake Champlain Basin Program, Grand Isle, VT.

Carriger, J. F., W. S. Fisher, T. B. Stockton, and P. E. Sturm. 2013. Advancing the Guánica Bay (Puerto Rico) Watershed Management Plan. Coastal Management **41**:19-38.

Cerosaletti, P. E., D. G. Fox, and L. E. Chase. 2004. Phosphorus reduction through precision feeding of dairy cattle. Journal of Dairy Science **87**:2314-2323.

Contach, K., C. Ballard, W. Emerich, C. Sniffen, and E. Thomas. 2003. The Feeding of Supplemental Phosphorus on Dairy Farms in the Lake Champlain Basin: An Education/Demonstration Project. Technical Report No. 42. Lake Champlain Basin Program, Grand Isle, VT.

Dou, Z. X., K. F. Knowlton, R. A. Kohn, Z. G. Wu, L. D. Satter, G. Y. Zhang, J. D. Toth, and J. D. Ferguson. 2002. Phosphorus characteristics of dairy feces affected by diets. Journal of Environmental Quality **31**:2058-2065.

Eberhard, R., C. J. Robinson, J. Waterhouse, J. Parslow, B. Hart, R. Grayson, and B. Taylor. 2009. Adaptive management for water quality planning - from theory to practice. Marine and Freshwater Research **60**:1189-1195.

Field, R., and E. J. Struzeski, Jr. 1972. Management and Control of Combined Sewer Overflows. Journal (Water Pollution Control Federation) **44**:1393-1415.

Ford, R. F. 2012. Agricultural Phosphorus in Vermont's Missisquoi Bay Watershed: History, Status, and Solutions. University of Vermont, Burlington, VT.

Gaddis, E. J. B., and A. Voinov. 2010. Spatially Explicit Modeling of Land Use Specific Phosphorus Transport Pathways to Improve TMDL Load Estimates and Implementation Planning. Water Resources Management **24**:1621-1644.

Gburek, W. J., E. Barberis, P. M. Haygarth, B. Kronvang, and C. Stamm. 2005. Phosphorus Mobility in the Landscape *in* J. T. Sims and A. N. Sharpley, editors. Phosphorus: Agriculture and the Environment. American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Pubs., Madison, WI.

Geohring, L. D., T. S. Steenhuis, A. S. Brooks, M. N. Rosenwald, J. Chen, and V. J. Putman. 1999. Cost-Effective Phosphorus Removal from Secondary Wastewater Effluent

through Mineral Adsorption. Technical Report No. 36. Lake Champlain Basin Program, Grand Isle, VT.
Ghebremichael, L. T., T. L. Veith, and M. C. Watzin. 2010. Determination of Critical Source Areas for phosphorus loss: Lake Champlain Basin, Vermont. Transactions of the Asabe **53**:1595-1604.
Gigerenzer, G. 2014. Risk Savvy: How to make good decisions. Penguin.
Goodwin, P., and G. Wright. 2004. Decision Analysis for Management Judgment: Fourth Edition. John Wiley & Sons, Ltd., West Sussex, England.
Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. Structured Decision Making: A Practical Guide to Environmental Management Choices. Wiley-Blackwell.
Gregory, R., D. Ohlson, and J. Arvai. 2006. Deconstructing adaptive management: Criteria for applications to environmental management. Ecological Applications **16**:2411-2425.
Gregory, S., A. W. Allen, M. Baker, K. Boyer, T. Dillaha, and J. Elliot. 2007. Realistic expectations of timing between conservation and restoration actions and ecological responses *in* M. Schnepf and C. Cox, editors. Managing Agricultural Landscapes for Environmental Quality: Strengthening the Science Base. Soil and Water Conservation Society, Ankeny, IA.
Gunderson, L., C. S. Holling, and S. S. Light, editors. 1995. Barriers and Bridges to the Renewal of Ecosystems and Institutions. Columbia University Press, New York.
Hammond, J. S., R. L. Keeney, and H. Raiffa. 1998. Even Swaps: A Rational Method for Making Trade-offs. Harvard Business Review.
Hegman, W., D. Wang, and C. Borer. 1999. Estimation of Lake Champlain Basinwide Nonpoint Source Phosphorus Export. Technical Report No.31. Lake Champlain Basin Program, Grand Isle, VT.
Hoffmann, J. P., E. A. Cassell, J. C. Drake, S. N. Levine, D. W. Meals, and D. Wang. 1996. Understanding Phosphorus Cycling, Transport and Storage in Stream Ecosystems as a Basis for Phosphorus Management. Technical Report No. 20. Lake Champlain Basin Program, Grand Isle, VT.
Holling, C. S., editor. 1978. Adaptive Environmental Assessment and Management. John Wiley & Sons, Chichester, New York, Brisbane, Toronto.
Hughes, J. W., W. E. Jokela, D. Wang, and C. Borer. 1999. Determination and Quantification of Factors Controlling Pollutant Delivery from Agricultural Land to Streams in the Lake Champlain Basin. Technical Report No. 35. Lake Champlain Basin Program, Grand Isle, VT.
Jensen, F. V., and T. D. Nielsen. 2007. Bayesian Networks and Decision Graphs. Springer, Berlin.
Jin, S. M., L. M. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment **132**:159-175.
Johnson, C. 1998. The Nature of Vermont. University Press of New England, Lebanon, NH.
Johnson, F., and K. Williams. 1999. Protocol and practice in the adaptive management of waterfowl harvests. Conservation Ecology **3**:article 8. [online] URL: http://www.ecologyandsociety.org/vol3/iss1/art8/.

Jokela, W. E., J. P. Tilley, and D. S. Ross. 2010. Manure Nutrient Content on Vermont Dairy Farms: Long-Term Trends and Relationships. Communications in Soil Science and Plant Analysis **41**:623-637.

Keeney, R. L. 1992. Value-focused thinking: a path to creative decision making. Harvard University Press, Boston, MA.

Kirkwood, C. W. 1992a. Estimating the impact of uncertainty on a determinisitc multiattribute evaluation. Management Science **38**:819-826.

Kirkwood, C. W. 1992b. An overview of methods for applied decision-analysis. Interfaces **22**:28-39.

Kline, M., and B. Cahoon. 2008. Restoring Ecological Processes in Protected River Corridors. River Management Program, Vermont Agency of Natural Resources, Waterbury, VT.

Lake Champlain Basin Program. 1996. Opportunities for Action: An Evolving Plan for the Future of the Lake Champlain Basin. Lake Champlain Basin Program, Grand Isle, VT.

Lake Champlain Basin Program. 2000. Preliminary Evaluation of Progress Toward Lake Champlain Basin Program Phosphorus Reduction Goals. Lake Champlain Basin Program, Grand Isle, VT.

Lake Champlain Basin Program. 2003. Opportunities for Action: An Evolving Plan for the Future of the Lake Champlain Basin. Lake Champlain Basin Program, Grand Isle, VT.

Lake Champlain Basin Program. 2010. Opportunities for Action: An Evolving Plan for the Future of the Lake Champlain Basin. Lake Champlain Basin Program, Grand Isle, VT.

Lake Champlain Basin Program. 2012. Publications Database Website. URL: http://www.lcbp.org/publication_search.aspx. Last Accessed: 01/18/2012.

Lake Champlain Study. 1979. Shaping the Future of Lake Champlain. Final Report. Burlington, VT.

Langendoen, E. J., A. Simon, L. Klimetz, N. Bankhead, and M. E. Ursic. 2012. Quantifying Sediment Loadings from Streambank Erosion in Selected Agricultural Watersheds Draining to Lake Champlain. Technical Report No. 72. Lake Champlain Basin Program, Grand Isle, VT.

LCBP. 2008. State of the Lake and Ecosystem Indicators Report. Lake Champlain Basin Program, Grand Isle, VT.

LCBP. 2012. State of the Lake and Ecosystem Indicators Report. Lake Champlain Steering Committee, Lake Champlain Basin Program, Grand Isle, VT.

Levine, S. N., A. D. Shambaugh, S. E. Pomeroy, and M. Braner. 1997. Phosphorus, nitrogen, and silica as controls on phytoplankton biomass and species composition in Lake Champlain (USA-Canada). Journal of Great Lakes Research **23**:131-148.

Maguire, R. O., G. L. Mullins, and M. Brosius. 2008. Evaluating long-term nitrogen- versus phosphorus-based nutrient management of poultry litter. Journal of Environmental Quality **37**:1810-1816.

Marcot, B. G. 1998. Selecting Appropriate Statistical Procedures and Asking the Right Questions: A Synthesis *in* V. Sit and B. Taylor, editors. Statistical Methods for Adaptive Management Studies. B.C. Ministry of Forests, Research Branch, Victoria, B.C.

Martin, J., P. L. Fackler, J. D. Nichols, M. C. Runge, C. L. McIntyre, B. L. Lubow, M. C. McCluskie, and J. A. Schmutz. 2011. An Adaptive-Management Framework for Optimal Control of Hiking Near Golden Eagle Nests in Denali National Park. Conservation Biology **25**:316-323.

McCann, R. K., B. G. Marcot, and R. Ellis. 2006. Bayesian belief networks: applications in ecology and natural resource management. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere **36**:3053-3062.

McCarthy, M. A., and H. P. Possingham. 2007. Active adaptive management for conservation. Conservation Biology **21**:956-963.

McDowell, R. W., A. N. Sharpley, and A. T. Chalmers. 2002. Land use and flow regime effects on phosphorus chemical dynamics in the fluvial sediment of the Winooski River, Vermont. Ecological Engineering **18**:477-487.

Meals, D. W. 1996. Watershed-scale response to agricultural diffuse pollution control programs in Vermont, USA. Water Science and Technology **33**:197-204.

Meals, D. W. 2001. Water quality response to riparian restoration in an agricultural watershed in Vermont, USA. Water Science and Technology **43**:175-182.

Meals, D. W., and L. F. Budd. 1998. Lake Champlain Basin nonpoint source phosphorus assessment. Journal of the American Water Resources Association **34**:251-265.

Meals, D. W., S. A. Dressing, and T. E. Davenport. 2010. Lag Time in Water Quality Response to Best Management Practices: A Review. Journal of Environmental Quality **39**:85-96.

Medalie, L., R. M. Hirsch, and S. A. Archfield. 2012. Use of flow-normalization to evaluate nutrient concentration and flux changes in Lake Champlain tributaries, 1990–2009. Journal of Great Lakes Research.

Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society **24**:602-612.

Michaud, A. R., I. Beaudin, J. Deslandes, F. Bonn, and C. A. Madramootoo. 2007. SWAT-predicted influence of different landscape and cropping system alterations on phosphorus mobility within the Pike River watershed of south-western Quebec. Canadian Journal of Soil Science **87**:329-344.

Mihuc, T. B., F. Dunlap, C. Binggeli, L. Myers, C. Pershyn, A. Groves, and A. Waring. 2012. Long-term patterns in Lake Champlain's zooplankton: 1992–2010. Journal of Great Lakes Research.

Morgan, M. G., and M. Henrion. 1990. Uncertainty. Cambridge University Press, Cambridge.

National Research Council. 2001. Nutrient Requirements of Dairy Cattle: Seventh Revised Edition.*in* Subcommittee on Dairy Cattle Nutrition Committee on Animal Nutrition, editor. National Academies Press, Washington, D.C.

Nichols, J. D., M. C. Runge, F. A. Johnson, and B. K. Williams. 2007. Adaptive harvest management of North American waterfowl populations: a brief history and future prospects. Journal of Ornithology **148**:S343-S349.

Owens, P. N. 2009. Adaptive management frameworks for natural resource management at the landscape scale: implications and applications for sediment resources. Journal of Soils and Sediments **9**:578-593.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics **32**:333-365.

Peterson, G. D., G. S. Cumming, and S. R. Carpenter. 2003. Scenario planning: a tool for conservation in an uncertain world. Conservation Biology **17**:358-366.

Phillips, P., and A. Chalmers. 2009. Wastewater effluent, combined sewer overflows, and other sources of organic compounds to Lake Champlain. Journal of the American Water Resources Association **45**:45-57.

Pulwarty, R. S., and T. S. Melis. 2001. Climate extremes and adaptive management on the Colorado River: Lessons from the 1997-1998 ENSO event. Journal of Environmental Management **63**:307-324.

Regan, H. M., M. Colyvan, and M. A. Burgman. 2002. A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecological Applications **12**:618-628.

Schreiber, E. S. G., A. R. Bearlin, S. J. Nicol, and C. R. Todd. 2004. Adaptive management: a synthesis of current understanding and effective application. Ecological Management & Restoration **5**:177-182.

Sharpley, A., and H. Tunney. 2000. Phosphorus research strategies to meet agricultural and environmental challenges of the 21st century. Journal of Environmental Quality **29**:176-181.

Slovic, P. 1987. Perception of risk. Science **236**:280-285.

Slovic, P. 1995. The construction of preference. American Psychologist **50**:364-371.

Slovic, P., B. Fischhoff, and S. Lichtenstein. 1977. Behavioral decision-theory. Annual Review of Psychology **28**:1-39.

Smeltzer, E. 1997. A Phosphorus Budget, Model, and Load Reduction Strategy for Lake Champlain. Submitted to US Environmental Protection Agency Region I and Region II.

Smeltzer, E., F. Dunlap, and M. Simoneau. 2009. Lake Champlain Phosphorus Concentrations and Loading Rates, 1990-2008. Technical Report No. 57. Lake Champlain Basin Program, Grand Isle, VT.

Smeltzer, E., A. D. Shambaugh, and P. Stangel. 2012. Environmental change in Lake Champlain revealed by long-term monitoring. Journal of Great Lakes Research **38, Supplement 1**:6-18.

Smeltzer, E., and M. Simoneau. 2008. Phosphorus Loading to Missisquoi bay from Sub-Basins in Vermont and Quebec, 2002-2005. Vermont Agency of Natural Resources and Ministere du Developpement durable, de l'Environnement at des Parcs.

Smil, V. 2000. Phosphorus in the environment: Natural flows and human interferences. Annual Review of Energy and the Environment **25**:53-88.

Stankey, G. H., R. N. Clark, and B. T. Bormann. 2005. Adaptive Management of Natural Resources: Theory, Concepts, and Management Institutions. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station,, Portland, OR.

Stickney, M. 2008. Building bridges, fording streams, reaching agreement in the Lake Champlain basin: Alternatives to legislation and regulation rooted in citizen and science-based approaches to inspire watershed protection. Water Sa **34**:468-475.

Stickney, M., C. Hickey, and R. Hoerr. 2001. Lake Champlain Basin Program: Working together today for tomorrow. Lakes & Reservoirs: Research & Management **6**:217-223.

Tetra Tech. 2013. Lake Champlain Basin SWAT Model Configuration, Calibration, and Validation. Prepared for U.S. EPA Region 1 - New England. Boston MA. Available from http://www.epa.gov/region1/eco/tmdl/lakechamplain.html.

Troy, A., D. Wang, D. Capen, J. O'Neil-Dunne, and S. MacFadden. 2007. Updating the Lake Champlain Basin Land Use Data to Improve Prediction of Phosphorus Loading. Technical Report No. 54. Lake Champlain Basin Program, Grand Isle, VT.

Tversky, A., and D. Kahneman. 1974. Judgement under Uncertainty: Heuristics and Biases. Science **185**:1124-1131.

U.S. Fish and Wildlife Service. 2011. Adaptive Harvest Management: 2011 Hunting Season. U.S. Deptartment of the Interior, Washington D.C. Available online at http://flyways.us/sites/default/files/uploads/2011_ahm_report.pdf.

United States Department of Agriculture. 2009a. 2007 Census of Agriculture: New York State and County Data. AC-07-A-32.*in* National Agricultural Statistics Service, editor. USDA.

United States Department of Agriculture. 2009b. 2007 Census of Agriculture: Vermont State and County Data. AC-07-A-45.*in* National Agricultural Statistics Service, editor. USDA.

Varis, O. 1997. Bayesian decision analysis for environmental and resource management. Environmental Modelling & Software **12**:177-185.

Vermont Agency of Natural Resources. 2010. A Framework for Remediation of Vermont's Stormwater-Impaired Waters. Vermont Department of Environmental Conservation, Waterbury, VT.

Vermont Agency of Natural Resources and Vermont Agency of Agriculture Food and Markets (VTANR & VTAAFM). 2011. Vermont Clean and Clear Action Plan 2010 Annual Report.

von Winterfeldt, D., and W. Edwards. 1986. Decision Analysis and Behavioral Research. Cambridge University Press, London.

Walling, D. E. 1999. Linking land use, erosion and sediment yields in river basins. Hydrobiologia **410**:223-240.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society **24**:706-723.

Walters, C. J. 1986. Adaptive Management of Renewable Resources. Macmillan Publishing Company, New York, NY.

Walters, C. J. 1997. Challenges in adaptive management of riparian and coastal ecosystems. Conservation Ecology **1**:1. [online] URL: http://www.consecol.org/vol1/iss2/art1/.

Walters, C. J. 2007. Is adaptive management helping to solve fisheries problems? Ambio **36**:304-307.

Watzin, M. C., S. Fuller, L. Bronson, R. Gorney, and L. Schuster. 2011. Monitoring and Evaluation of Cyanobacteria in Lake Champlain. Technical Report No. 61. Grand Isle, VT.

Watzin, M. C., R. L. Smyth, E. A. Cassell, W. C. Hession, R. E. Manning, and D. Wang. 2005. Ecosystem Indicators and an Environmental Score Card for the Lake Champlain Basin Program. Technical Report No. 46. Lake Champlain Basin Program, Grand Isle, VT.

Weller, C. M., M. C. Watzin, and D. Wang. 1996. Role of wetlands in reducing phosphorus loading to surface water in eight watersheds in the Lake Champlain Basin. Environmental Management **20**:731-739.

Wichelns, D. 2010. Embracing Uncertainty to Improve Water Management, with Examples from Seven River Basins. International Journal of Water Resources Development **26**:495-508.
Williams, B. K. 2011a. Adaptive management of natural resources-framework and issues. Journal of Environmental Management **92**:1346-1353.
Williams, B. K. 2011b. Passive and active adaptive management: Approaches and an example. Journal of Environmental Management **92**:1371-1378.
Williams, B. K., and E. D. Brown. 2012. Adaptive Management: The U.S. Department of the Interior Applications Guide. Department of the Interior Adaptive Management Working Group, Washington DC.
Winchell, M., D. W. Meals, S. Folle, J. Moore, D. C. Braun, C. DeLeo, and K. Budreski. 2011. Identification of Critical Source Areas of Phosphorus Within the Vermont Sector of the Missisquoi Bay Basin. Technical Report No. 63b. Lake Champlain Basin Program, Grand Isle, VT.
Winsten, J. R. 2004. Developing and Assessing Policy Options for Reducing Phosphorus Loading in Lake Champlain. Technical Report No. 47. Lake Champlain Basin Program, Grand Isle, VT.
Zuzel, J. F., and J. L. Pikul. 1987. Infiltration into a seasonally frozen agricultural soil. Journal of Soil and Water Conservation **42**:447-450.

CHAPTER 2. DEVELOPMENT OF AN INDICATOR DATABASE FOR DECISION-AIDING AND ADAPTIVE PHOSPHORUS MANAGEMENT IN THE LAKE CHAMPLAIN BASIN

Executive Project Summary

Phosphorus is widely regarded as the primary cause of consistent annual algae blooms in many parts of Lake Champlain, and as a result, reduced tributary phosphorus loads have become a primary indicator of management success. However, the existing monitoring programs in the Basin function primarily to track algae blooms and lower phosphorus loading rates, and so an ability to understand the observed lack of progress over the past 20 years is beyond the scope of the currently available data. Therefore, in addition to a lack of consistent progress toward phosphorus loading management goals, there is relatively little concrete information explaining why tributary loading rates have not decreased as expected, relative to management efforts to-date.

In 2009, as the LCBP Steering and Technical Advisory Committees began the third update of *Opportunities for Action* (OFA)*,* the LCBP's management plan, they expressed a desire to develop an adaptive management framework that could be applied to the phosphorus management initiatives outlined in OFA. In particular, the Steering Committee was interested in using an adaptive management approach to make further management progress while helping to shed light on the answers to several basic questions about the relationship between the management actions taken so far and improvements in water quality in the lake. Central to the adaptive management approach in the context of water quality are answers to other questions about which management actions are the most effective and the most cost-effective for achieving reductions in phosphorus loading, about what levels of phosphorus reduction could be achieved if the entire "universe of need" were to be managed, and about how filling major existing knowledge gaps could improve decision-making around which management initiatives to pursue.

To this end, we had four specific aims:

- to provide a method for tracking the implementation of commitments in *Opportunities for Action*, and any ecological response at a common watershed scale;
- to identify areas of strong opportunity for future management by quantifying the universe of need;
- to provide simple estimates of effectiveness and efficiency for each of the major management initiatives in the Lake Champlain basin; and
- to identify important knowledge gaps in our understanding of what management actions have occurred or in the effects of that management.

We used a performance-based indicator approach modeled loosely on Watzin et al.'s 2005 report for the LCBP (*Ecosystem Indicators and an Environmental Score Card for the Lake Champlain Basin Program*), developing indicators for each major management initiative defined in OFA. For each indicator, we attempted to track the current state, to set short and long term goals, to estimate reductions from achieving those goals, and to estimate cost-effectiveness of each initiative.

The data indicate that better management of agricultural crop and hayland and of runoff from impervious surfaces present the largest opportunities for management into the future. Wastewater treatment, farmstead management, and combined sewer overflows present comparatively small opportunities for achieving reductions on the scale required to make progress in much of Lake Champlain using existing regulatory tools. According to our estimates, 190 metric tons per year (mt/yr) of phosphorus reduction could be achieved through better management of crop and hayland, and 44 mt/yr could be reduced from managing stormwater runoff. Increased management of farmsteads, CSO abatement, backroad management, and better wastewater compliance could account for a combined 35 mt/yr.

These results suggest that those pollution sources that have defined or identifiable locations (whether they are classified as point or nonpoint sources) have been easier to manage, and that the much harder to manage sources are those that accumulate slowly and are more distributed across the landscape, such as exposed agricultural soil and streambanks.

The data also showed clearly that the cost to achieve the reductions vary widely by management initiative. For example, although agricultural field management constitutes by far the largest opportunity for reductions, its overall cost ($392 million) is nearly an order of magnitude lower than the cost to manage runoff from impervious surfaces ($2.3 billion). The cost estimates for farmstead BMPs and CSO abatement total $184 million, though their much lower potential for reductions points to the importance of considering both total potential and cost-effectiveness.

When considering the interaction between potential for reductions and the cost to achieve those reductions, the data show that the management of runoff from impervious surfaces is by far the least cost effective of any of the practices, with an average cost of ~$2200 per kg of P compared with ~$130 per kg of P for crop and hayland practices. Farmstead BMPs and backroad maintenance are similar to in cost-effectiveness to crop and hayland practices, though as noted above, they share an overall lower potential for large scale phosphorus reductions.

The major lessons from this work were:

- Major knowledge gaps still exist in understanding the watershed-level effects of local-scale management practices, and in the effectiveness of certain novel management policies and practices (particularly those policies and practices dealing with stream corridors). These knowledge gaps can only be addressed through targeted research and subsequent long term monitoring.
- The greatest potential for future phosphorus reductions lies in the most diffuse of the nonpoint sources – agricultural crop and hayland and stormwater runoff. We estimate that these two sources account for more than 85% of the total potential reductions.
- The cost of managing each of the major pollution sources varies widely watershed to watershed and across pollution source types. The variation between watersheds can be as much as a factor of 8 or 9 while the variation across source types can be as much as a factor of 100.
- These large differences in cost to manage each pollution sector point to important tradeoffs that the LCBP and its partners must make, such as those between cost-effectiveness and equal burden between pollution sectors, between implementation and research, and between relatively short- and very long-term solutions.

Introduction

Over the past several decades, algae blooms have become a consistent problem in parts of Lake Champlain, presenting impairments to recreation and occasionally causing fish kills. Recently, these blooms have become more dominated by cyanobacteria, causing additional public health concerns, including risk to drinking water intakes and further limiting public recreation. The results of early studies named excess phosphorus as the most likely driver of the increase in the occurrence of algae blooms and identified the most likely source as runoff and erosion from the Lake's watersheds (Lake Champlain Study 1979). These studies listed several watershed sources of phosphorus including wastewater treatment and non-point source loading from agricultural and urban land uses.

Initial management efforts targeted wastewater treatment to great effect, and as those projects were finished, concerted efforts to control nonpoint loading became more earnest. Due to the dispersed nature of nonpoint source phosphorus pollution, specific management actions have focused on reducing erosion from the landscape to streams, with the expectation that this will in turn reduce the occurrence of algae blooms in the Lake. As a result of this shift in management focus from algae blooms to phosphorus delivery to the lake, reduced tributary phosphorus loads have become a primary indicator of management success.

However, despite the substantial amount of time and money invested in trying to reduce phosphorus delivery to the lake, monitoring data have shown little change in reducing phosphorus loading or algae blooms. Because the existing monitoring programs in the Basin track progress relative to only the key management targets (in this case, only algae blooms frequency and severity and phosphorus loads), an ability to understand the observed lack of progress is beyond the scope of currently available data. Therefore, in addition to a lack of consistent progress toward management goals, there is relatively little concrete information explaining why tributary loading rates have not decreased as expected, relative to management efforts to-date.

In 1988, the states of Vermont, New York, and the province of Quebec signed a Memorandum of Understanding that initiated a process to establish in-lake criteria for phosphorus concentrations, and establish target watershed loadings to achieve the in-lake criteria (Stickney et al. 2001). The Lake Champlain Steering Committee was established to guide that process. The Lake Champlain Special Designation Act of 1990 created a group responsible for developing a comprehensive plan to prevent and control phosphorus pollution, with the goal of restoring Lake Champlain water quality. This group, called the Lake Champlain Management Conference, produced a plan and recommended that the Lake Champlain Steering Committee coordinate implementation of the plan, and all other efforts among the three jurisdictions for research, demonstration projects, lake and tributary monitoring, and education and outreach initiatives. The Lake Champlain Basin Program (LCBP) was established to staff the Management Conference and the Steering Committee, and it continues to assist with coordinated management of the lake's natural and cultural resources between Vermont, New York, and Quebec.

In 2009, as the LCBP Steering Committee and its Technical Advisory Committee began the third update of *Opportunities for Action* (OFA), the LCBP's management plan, several of the issues described above led to an interest in the development of tools to help learn about the effectiveness of various management actions and to increase the use of available monitoring and research data to guide the LCBP's decision-making processes. A lack of clarity about the effectiveness of various management initiatives, disagreements about how to prioritize funding allocations, and a desire for greater accountability also contributed to this interest. In late 2009, the LCBP Steering Committee formalized a desire to develop an adaptive management framework that could be applied to the phosphorus management initiatives outlined in OFA. In particular, the Steering Committee was interested in using an adaptive management approach to help shed light on the answers to several questions, including:

1. What is the relationship between the management actions taken so far and the "universe of need"? (i.e., how much has been done, and how much is left to do?)
2. Which management actions are the most environmentally effective and the most cost-effective for achieving reductions in phosphorus loading?
3. What levels of phosphorus reduction could be achieved if the entire "universe of need" were to be managed?
4. What major knowledge gaps exist, and how could filling those gaps improve decision-making?

Developing the answers to these questions comprise the bulk of a formal process of Adaptive Environmental Assessment and Management (AEAM), as described by Carl Walters (1986). These assessments, which provide the information critical to informing adaptive management processes, serve as knowledge-gathering exercises to understand what options exist for achieving management goals and what the likely effects of those actions are. Adaptive management, which follows these assessments, describes the use of a set of tools that helps resource managers address the knowledge gaps discovered in the assessment phase, and learn more about the effectiveness of their actions on the environment. As such, the goal of an Adaptive Environmental Assessment is to enable adaptive management in the future, which in turn helps resource managers to become more effective.

In its modern form, adaptive management describes a highly structured, well-planned cycle of "learning by doing" that uses decision analysis tools to make the best possible decision about management actions given the available information and then uses an experimental approach to strategically improve the quality of information used in making future decisions. The use of the tools of adaptive management come with a few assumptions about the kind of problem addressed with this sort of approach; the management decisions are recurring at some regular and predictable interval, that there are multiple stakeholders who hold multiple objectives for the outcome of the management actions, and that there is a high degree of uncertainty about the outcomes of management actions. While there is wide opportunity for interpretation in how many steps there should be, what they are called, and how they are divided or clumped, there is wide consensus that adaptive management must consist of iterations between the decision-making component and an opportunity for learning through inference (Allen et al. 2011).

For the tools of adaptive management to prove helpful, the foundations of two key processes must be present. The first is a process for making decisions that considers multiple objectives at the same time, evaluates action alternatives relative to each other, and produces repeatable and defensible results. The tools that enable this sort of process are referred to in general as Structured Decision-Making (SDM) methods (Gregory et al. 2012). SDM methods have been successfully used in a wide variety of situations, and increasingly are being used by the U.S. federal government in natural resources management as a way to make more informed and defensible decisions about the best use of public resources (Stankey et al. 2005, Williams and Brown 2012).

In the context of Lake Champlain, the use of SDM methods requires an understanding of what management goals are most appropriate from both a scientific and a public perspective – i.e., what do the data suggest are appropriate goals, and what goals do the public and other stakeholders want to see achieved. A large body of research has shown that explicitly including these sorts of values into decision-making improves the quality of the outcomes. However, to do that, decision-makers need to be clear about what value-based goals are important at the outset of the decision-making process. This information paves the way toward better generation of management options and better ability to make informed tradeoffs later in the decision process.

The second key process for successful application of adaptive management tools is a way to learn about management actions through inference; generally this is accomplished by using statistical methods to compare the predicted outcomes of management actions to their actual monitored outcomes (Walters 1986). Resource managers often rely on complex computer models to generate these predictions, but research and experience have shown that relatively simple estimations that use basic methods and existing data can often provide enough discrimination among alternatives to enable more informed decisions, with lower investments of time and money.

The tools of SDM and adaptive management can prove especially helpful when one or more of the following conditions is true: 1) a high degree of uncertainty about the structure and function of the ecological system exists, 2) where there are many stakeholders and multiple objectives for the relevant management agencies and 3) those agencies make recurring management decisions (either cooperatively or in parallel) about the same resource. All of these conditions are true of the Lake Champlain Basin. The use of these tools has proven especially successful in helping managers and scientists gain a better understanding of ecosystem function in other large and complex systems (Pulwarty and Melis 2001) and of how resources respond to management actions (Johnson and Williams 1999, Johnson et al. 2002).

Because adaptive management is a means for developing better understanding of complex interactions between management actions and the environment, model representations of these interactions are a major component of adaptive management efforts. Not surprisingly, many of these representations are complex computer models that require a large amount of training to build, understand, and operate. These complex models are often very useful for shedding light onto environmental phenomena and for predicting the response of the ecosystem to management interventions and natural events (NRC 2007). However, the large number of parameters and interactions in these sorts of models make them subject to large uncertainties in their predictions, which are often difficult to quantify or even identify (NRC 2007). In the context of managing water resources, these uncertainties are particularly problematic for setting targets and designing management strategies for restoration plans, such as in the EPA's Total Maximum Daily Load (TMDL) program.

In response to these large uncertainties, there has been a growing effort to develop simpler models that lend themselves well to statistical methods to quantify the precision and accuracy of the model predictions (Caulkins 2002), particularly for models used in TMDL assessment and implementation phases (Reckhow 2003, Shirmohammadi et al. 2006). In some cases, these are empirical statistical models, but there is also increasing use of various forms of system-oriented models that are able to quantify diverse kinds of relationships in situations where data are limited and in ways that are often more relevant for policy development. These sorts of models are variously called Cognitive Maps, Causal Maps, Analytic Network Process Models, or Influence Diagrams, but share the common trait that nodes representing the state of any variable are linked via arrows that represent causal connections (Figure 2-1, from (Watzin et al. 2005)). Depending on the kind of model and its purpose, the nodes and arrows can represent real or estimated quantities or they can represent qualitative relationships. One advantage of these sorts of models is that they are easily translatable into sets of indicators of important ecological or management conditions.

A substantial amount of research has explored the use of indicators in the conservation and resource management world as a method for quantifying vague and amorphous concepts such as "ecosystem health" and the effect of management on ecosystems (EPA 2000), including work done in the Lake Champlain Basin (Watzin et al. 2005).

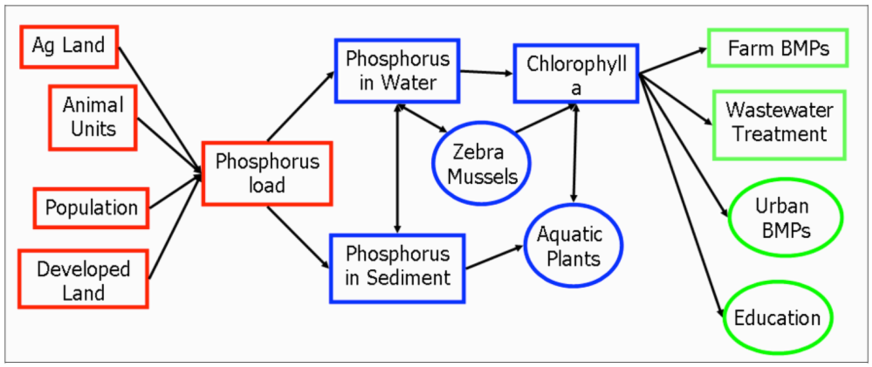

Figure 2-1. Conceptual model diagram showing dependencies (arrows) between different system elements (circles and squares); Watzin et al. 2005.

Project Goals

The overarching aim of this project was to lay the groundwork to enable the development of an adaptive management framework – that is, to enable the use of formalized SDM tools, and to generate predictions about management effectiveness and provide a method for comparing them to observations in the future. Our intention was to tabulate data that could be revised over time and that would be used as inputs to a more formal decision-making process developed separately from this effort.

There were four specific objectives we explored in the pursuit of enabling an adaptive management approach for the LCBP:

1. Provide a method for tracking the implementation of commitments in *Opportunities for Action*, and an ecological response at a common watershed scale,

2. Identify areas of strong opportunity for future management by quantifying the universe of need,
3. Provide simple estimates of effectiveness and efficiency for each of the major management initiatives tracked in the Indicator table, and
4. Identify important knowledge gaps in our understanding of what management actions have occurred or in the effects of that management.

To achieve these objectives we adopted a group of performance indicators to guide the data collection and analysis phases of this project. One of the key goals in the LCBP's developing adaptive management effort is to relate management progress to changing ecological condition. Quantitative measures of vague concepts such as "management progress" and "ecological condition" require the use of more specific, often stand-in, indicator variables for the issues of real interest. Performance indicators (referred to below as simply "indicators") fill this and several other key functions for informing the use of formal decision making tools and in enabling learning over time. Specifically, the ability of indicators to quantify specific and concrete components of broad management goals means that they can enable clear connections between the available management options (i.e., policy instruments) and their supposed ecological effect (Wolfslehner and Vacik 2011). These features of indicator systems in turn lend themselves well to the development of conceptual and quantitative models that can be used as part of adaptive management efforts.

Methods:

We opted for a performance indicator-based approach that would allow specific and quantitative measures of both management progress and ecological condition, and that would enable the development of hypotheses about how certain indicators were linked. We divided the group of indicators into two categories; one that tracks the implementation of major management programs detailed in OFA (Implementation Indicators) and a second that tracks changes in various components of the ecological condition of the Lake Champlain Basin (Ecosystem State Indicators). For each of these indicators, our goal was to characterize the "Current State", or the best estimate of the current value using best available data. Quantitative short- and long-term goals for each implantation indicator were developed along with expected phosphorus reductions that would result from achieving those goals, costs, and a measure of cost-effectiveness for each implementation indicator (see full tables in Appendix A).

Development of the Indicators

The phosphorus management chapter of Opportunities for Action organizes management tasks and commitments into major land-use pollution sectors, including agricultural lands, developed lands, rural lands and backroads, and floodplains, wetlands, and riparian areas. Within each of these sectors, the major existing phosphorus control initiatives are described by the individual commitments made by each LCBP partner in OFA. We organized those commitments that detailed specific implementation actions (as opposed to, for example, maintaining partnerships) into thematic groups that informed the development of the set of specific Implementation indicators within each land use sector (Table 2-1).

In 2010, an Adaptive Management workgroup, which comprises a subset of the LCBP Technical Advisory Committee and includes representatives from each jurisdiction, began meeting regularly to refine the language describing each indicator and to identify existing datasets that could be used to characterize the indicators. This effort was aligned very closely with a parallel effort at the Vermont Agency of Natural Resources Ecosystem Restoration Program, and so many of the many of the resulting indicators bear strong similarities to indicators developed as part of that effort. We departed from that effort slightly in that wherever possible, we attempted to replace jurisdiction-specific language or initiatives with language that was more broadly applicable across the basin.

These Implementation indicators were then related to a set of Ecosystem State indicators (Table 2-2), which track key ecosystem elements such as land use and land cover, stream channel condition, phosphorus load from various land uses, and total tributary phosphorus load. The basis for the selection of these ecosystem elements was to try to identify variables that are more proximately influenced by management decisions than are end-of-tributary phosphorus loads, where effectiveness has been measured to date. As existing

Box 1. Translating Commitments in Opportunities for Action into measureable Implementation and Ecosystem State Indicators:

The Agencies of Agriculture for each jurisdiction have committed to ensuring that all farms falling under relevant regulation (i.e., EPA for Vermont and New York, and MDDEFP for Quebec) have the necessary structures to prevent phosphorus pollution from four locations on the farmstead – manure pits, silage bunkers, milkhouse waste, and runoff from the barnyard (OFA tasks 4.1.14, 4.1.15, 4.1.19, 4.1.20, & 4.1.21). For example:

<u>OFA task 4.1.20:</u> Ensure that all (118) MFO farms in the Basin have the necessary structures in the production area needed to prevent direct farmstead discharges by 2013 (based on the number of farms available as of 2009).

From these four commitments we generated an Implementation indicator that tracked the percentage of farms that have and maintain those structures:

Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly maintained Best Management Practice structures, by structure type & farm size.

This Implementation indicator is paired with a corresponding Ecosystem State indicator that, as a result of farmstead management, would be expected to change in value:

Estimated P loss (mt/yr) from farmsteads.

management policies are applied more widely and new management policies are developed, changes in some of these variables (such as soil P levels) may become apparent before changes are seen in end-of-tributary loads (see box 1 for an example of how we translated commitments in OFA into measurable Implementation and Ecosystem State Indicators).

Calculation of Current States

One of the basic questions that we attempted to address though this effort was how much phosphorus management has occurred, in relation to the level of management that could be done (i.e., its "universe of need"). The goal was not to provide a complete census of all management actions in the sense of counting every square foot of managed impervious surface or every manure pit, but instead to get a general sense of how much effort had been expended in controlling each pollution source described in the indicator list.

In order to relate this information in one common language, we expressed the current state of each Implementation indicator relative to its universe of need, which provided some insight into where large opportunities for management still exist, and which sources have been managed at or close to the maximum level. To do this, we expressed each indicator current state as a percentage of what could be achieved. For example, the acreage of agricultural land currently cover cropped was divided by the acreage of cropland (which excludes farmstead footprints, pasture, and hayland, where cover crops could not be used). Though the approximations made in this method are relatively crude and subject to some uncertainty, it does provide a general sense of the relative possibility for expansion of each management initiative in the Lake Champlain Basin.

Table 2-1. Implementation Indicators sorted by land use sector

Agricultural Lands
• Percent of agricultural land under enhanced land management for: a. Cover cropping b. Alternative manure spreading methods c. Conservation tillage • Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO, or Large/Medium CAFOs) • Percent of farms operating within 5% of whole-farm P balance • Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by structure type and farm size: a. Manure storage b. Silage leachate treatment c. Barnyard runoff treatment d. Milkhouse waste treatment • Percent of farm inspections identifying substantial violations of relevant agricultural regulation • Percent of perennial stream miles where livestock have uncontrolled access to the stream
Developed Lands
• Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit • Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit • Percent of all permitted operational stormwater sites in substantial compliance with their permit • Percent of municipalities with storm sewer systems that have completed IDDE projects • Percent of impervious area that is under stormwater management • Number of combined sewer outfalls remaining in the Lake Champlain Basin • Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment • Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate. • Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin
Rural Lands/Backroads
• Percent of sampling units within logging jobs in the Vermont and New York portions of the Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams. • Percent of Vermont towns participating in the Better Backroads Program (or equivalent program) • Percent of towns that have completed road erosion needs inventories and capital budget plans • Percent of priority erosion control projects identified in road erosion needs inventories that are completed
River, Floodplain, and Wetland Conservation & Restoration
• Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply. • Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored • Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs • Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances • Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved • Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans
Wastewater
• Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations • Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system

Table 2-2. Ecosystem State Indicators

Ecosystem State Indicators:
• Median animal units per acre • Ratio of imported P / exported P on agricultural lands • Average total P loss from cropland (including hay) (mt/ha/yr) • Average total P loss from farmsteads (mt/ha/yr) • Ratio of imported P / exported P on urban lands • Average total P loss from urban areas (mt/ha/yr) • Average total P loss from road network (mt/ha/yr) • Mean soil P level in cropland (includes rotated and permanent hay) • Mean soil P level in pastureland • Best recent estimates for percent of land in the following categories: a. annual crops b. hay, pasture, lawn c. impervious surface • Percent of river miles in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) • P applied to developed lands (mt/ha/yr) • 5-year average wastewater phosphorus load (2007-2011) (mt/yr) • 5-year average non-point source phosphorus load (2007-2011) (mt/yr) • 5-year average tributary total phosphorus load (2007-2011) (mt/yr) • 6-year ratio of dissolved P : total P in tributary loads (2007-2012)

The datasets we used to calculate the current state values for the set of Implementation indicators were delivered directly from LCBP partner agencies over the course of 2012, and reflected the best available information at the time. In most cases, the data were summarized directly from record-keeping databases, aggregated by watershed or town. We then summarized these aggregated data by major tributary basin, which corresponds roughly to the USGS Hydrologic Unit Code 8-digit (HUC 8) watershed boundaries (Figure 2-2). In a few cases, data were summarized at the state level and no finer-scale divisions were possible. These state-level data were generally extracted from agency annual reports. Simple summations were very often sufficient to characterize the data by watershed. Exceptions to this generalization are noted in the Indictor Table itself, and explained in Appendix B of this report, which details indicator-by-indicator calculation notes. It should be noted that many of these data sources are in constant revision. While the data used in this report reflected the best available data at the time of writing, many of the datasets have subsequently been revised, and therefore the data presented here may not reflect the most current version of any particular dataset.

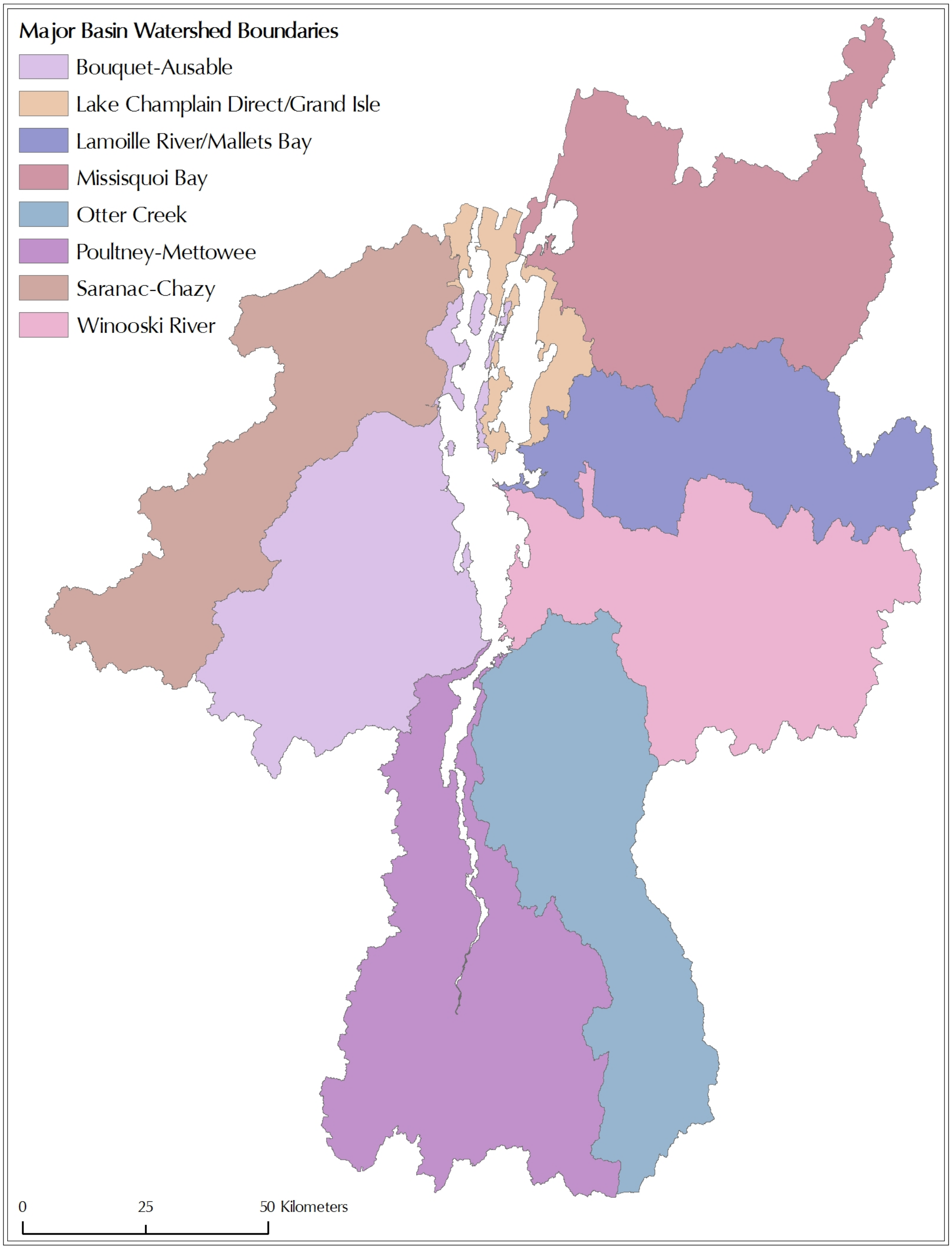

Figure 2-2. Major 8-digit Hydrologic Unit Code (HUC 8) tributary basins of Lake Champlain used to summarize spatially-explicit management data by basin.

Data to populate the current state values of the Ecosystem State indicators came from a variety of existing datasets. The modeling effort associated with the ongoing revision of the Lake Champlain Phosphorus TMDL provided data for indicators estimating land cover and estimating phosphorus loads from different land uses. Land-use based phosphorus loads were aggregated within watersheds from all similar sources to estimate the four land use loading estimates called for in the indicator table (i.e., cropland, farmsteads, urban areas, road network). These land-use specific estimates, which were based on long-term averages, were used as the basis for estimating the phosphorus reductions discussed later in this report. Land cover estimates came from the land use layer developed as part of the same modeling effort. The land use layer developed by Tetra Tech used the National Land Cover Database (NLCD) 2006 version as a base layer, but then augmented that layer with a variety of data, including specific crop data from the 2008 Cropland Data Layer, soils data from the USDA SSURGO soils database, road and driveways locations from VTrans and the E911 GIS layers, and from NRCS for locations of farmsteads.

We estimated tributary phosphorus loads using the total phosphorus (TP) and dissolved phosphorus (DP) data from the Vermont Long Term Monitoring Program and the Weighted Regression on Time, Discharge, and Season (WRTDS) methods developed by Robert Hirsch et al. (2010) and used recently for Lake Champlain by Laura Medalie et al. (2012). The reported total phosphorus loads are the standard estimates from that method, which are similar in nature to a USGS LOADEST estimate. Bias statistics for these estimates equaled or bettered those reported by Medalie in her recent report. Following the method used by Vermont DEC for generating tributary load estimates, the estimates of phosphorus load at the flow monitoring gauge were adjusted upward to reflect the load at the true mouth of the tributary by using the ratio of area of land upstream of the gauge relative to the area of the full watershed. To estimate the phosphorus load for the Lake Champlain Direct/Grand Isle watershed (which is not gauged or sampled), we used a similar method where the proportion of the direct drainages area relative to the total area of the gauged watersheds was applied to the total phosphorus loads from all gauged watersheds. Non-point loads are the difference between the wastewater load for each watershed and the total load for that watershed.

We calculated ratios of dissolved phosphorus (DP) to total phosphorus (TP) from estimates of daily fluxes produced by the WRTDS method for each form of phosphorus. We summed these daily fluxes by season (within years), where “fall” is the first three months of the water year (October, November, December), “winter” is January, February, and March, and so on, and calculated a ratio of the DP and TP fluxes for each season, and then averaged these values within years. We opted for this seasonal averaging method because we felt that simple ratios of daily flux estimates over-emphasized the role of DP in the winter (which is relatively higher at that time of year) and a ratio of annual flux estimates erased too much of the variability that occurs over the course of the year.

Watershed-specific estimates of stream geomorphic evolution stage were taken from results from the Vermont River Management Program’s Phase 2 Stream Geomorphic Assessment program. The proportions reported in the table are the proportion of stream reaches in evolution stage II or III to all stream reaches assessed.

The area of impervious surface in each basin was summed within HUC 8 boundaries from the recent impervious surface layer created for the LCBP by the University of Vermont Spatial Analysis Lab (O'Neil-Dunne 2013). The base data for that project was 2011 orthophotography, and auxiliary datasets to identify roads and driveways.

Short- and Long-Term Acceptable Levels

A second aim of this project was to provide a method for relating management actions to short and long-term goals, and to tie those goals to hypothesized phosphorus reductions. In the context of this effort, the explicit short- and long-term goals (called "Acceptable Levels" in the indicator table) represented two different kinds of goals. The long-term goals represent the level of management that, according to best professional judgment, is necessary to achieve the desired ecological outcomes – in this case, reduced algae blooms. How and where these long-term goals are set is reflective of the scientific opinion of the Adaptive Management Workgroup more than of any programmatic or policy considerations. In the current iteration of the Indicator table, these goals were set at the entire universe of need, but as the management community learns more about the effectiveness of management practices, these levels could and should be revised as necessary.

In contrast to the long-term goals, the short-term goals are reflections of what is politically and fiscally feasible in the short-term – they are therefore policy decisions, and not based in scientific opinion. In the current table, these levels are taken from the commitments in OFA relating to each indicator, but other targets could be used as appropriate.

Short- and Long-Term Expected Phosphorus Reductions

A key element of good decision-making is an ability to compare the outcomes of various alternatives in light of each other. In the context of water quality management, that means making clear statements about the expected benefits and costs of various management initiatives (e.g., managing stormwater versus managing farmsteads). Since phosphorus loading is the main (direct) target for these management efforts, estimated phosphorus reductions were the sole benefit considered. We acknowledge that phosphorus loading is not the only important consideration, but the effectiveness of management practices should be a key driver in future decisions about management policies. These estimates, or hypotheses, were the main method by which we attempted to link the Implementation indicators to the Ecosystem State indicators.

To estimate short-term and long-term expected reductions, we used reduction efficiency values reported in the scientific and technical literature that were reported for similar management practices, and for similar climates. We applied those efficiency values (often a percentage) to the Tetra Tech estimate of the phosphorus load associated with the appropriate land use (Tetra Tech 2013a). This estimated phosphorus reduction reflects what could be achieved by implementing that practice on 100% of a particular land use. We then multiplied this reduction estimate by the proportion of that land use that could theoretically receive treatment; for our purposes, this was equivalent to the difference between the current state and the short- and long-term goals (see Box 2 for an example). Note that at the time of writing, sediment and nutrient loads provided in the model did not take into account transport loss,

Box 2. Estimating Potential Phosphorus Reductions:
In the Winooski Basin, VT AAFM reported 918 acres of cover crop for 2012, applied to the 61,274 acres of crop and hay land (estimated by Tetra Tech, 2013) in the watershed. This translates to 1.5% of the total productive land. The ultimate goal, for example, would be to cover crop all annual cropland, which represents 29.2% of the total crop and hay acres. The "current state" is 1.5%, the "ultimate acceptable level" is 29.2%.

Tetra Tech estimates the average TP load from cropland (including hay) to be 28.5 mt/yr in the Winooski watershed. Michaud et al. (2002) report a 30% reduction in TP from wide-scale cover cropping. The estimated reduction from cover cropping is calculated this way:

(Reduction rate * Land use load) * (Remaining opportunity)
= Expected reduction

(30% * 28.5 mt/yr) * (29.2% - 1.5%) = 2.4 mt/yr and therefore may have over-estimated delivered sediment and nutrient loads to the lake. Given this information, estimates of reductions provided in this report based on the TetraTech SWAT model might be high.

Total Cost and Cost-Effectiveness

To accomplish the third major goal of this effort, to provide simple estimates of effectiveness and efficiency for each indicator, we estimated two separate measures of cost associated with achieving the ultimate acceptable management level as specified in the Indicator Table. The first measure attempted to quantify total up-front investments to achieve the long-term acceptable levels. Many of the management practices described in the indicator table (Table 2-1) require heavy initial investments in construction and engineering in addition to yearly operation and maintenance (O&M), and spreading the total cost of the these practices over the lifespan of the practice can mask the (often substantial) initial investments required. We therefore included construction costs and engineering and design costs (D&E), but excluded program administration and O&M costs. We also excluded land costs because of the extreme variability of land prices around the basin and over time, and because of the vastly different amounts of land required for each type of practice.

However, because we were interested in more direct comparisons between management policies that require heavy initial investments (such as stormwater management) and those policies that are annual costs (such as agricultural field management), we also developed a 20-year cost estimate that included all of the costs described above, in addition to annual O&M costs. We assumed 20-year lifespans for urban stormwater practices (Schueler et al. 2007), 10-year lifespans for farmstead structural Best Management Practices (BMPs) (Gitau et al. 2006) and rural road, or backroad, BMPs (Garton 2013), and single-year lifespans for agricultural field practices. To calculate the 20-year cost, we added initial construction investments to the annual O&M costs over the lifetime of the practice. This amount was then multiplied by the number of times the practices would be replaced over 20 years, again excluding land costs.

We did not assume any diminishing (or increasing) marginal returns for the cost of different levels of management, which a more detailed economic analysis might. Economies of scale do exist for wide-scale stormwater construction efforts (Schueler et al. 2007), but a detailed assessment of the economics of watershed-scale phosphorus management was not one our intentions, and is beyond the scope of this project.

Our estimate of cost-effectiveness used the annualized 20-year cost and the long-term effectiveness to develop a ratio of the total cost to the expected phosphorus reductions when the long-term goal has been achieved for a particular BMP. This ratio was expressed in dollars per year for each kilogram of phosphorus reduced (i.e., dollars per year per kilogram per year), but can also be interpreted as dollars per kilogram of phosphorus reduced. This metric allowed direct comparisons between management policies that require heavy initial investments with those that require steady annual costs.

Results

Goal 1: Provide a method for tracking the implementation of commitments in OFA, and any ecological response

There were large differences in the amount of available tracking data for implementation efforts both between and within jurisdictions. As a result, we were unable to characterize the current states for every indicator we developed – there were data for 75% of the Implementation indicators (24 out of 32) and 72% of the Ecosystem State indicators (13 out of 18) for the Vermont watersheds, including the Quebec portion the Missisquoi.

Because of some of the differences in the existence of certain programs between Vermont and Quebec some of the data used in the calculations comes from only one jurisdiction. For the New York side of the basin, data existed for 30% of the Implementation indicators (10 out of 32) and 45% of the Ecosystem State indicators (8 out of 15).

Of those indicators for which data did exist, most did not have a clearly quantified short-term acceptable level detailed in OFA. This lack of goal-setting made the calculation of expected phosphorus reductions impossible for the short-term. However, because the Adaptive Management Workgroup did set long-term acceptable levels, we were able to calculate reductions for many of these. In the calculation of the long-term reductions, a primary limiting factor was a lack of data for management initiatives that are not common nationally. Very few of the effectiveness data we used were locally produced, and as a result, the expected reductions for practices that are not common water quality management practices nationally were difficult to quantify (e.g., maintenance of backroads – though a common management practice, it's not often thought of as a phosphorus management tool). All in all, we were able to estimate potential phosphorus reductions for 18 of the 32 indicators for the Vermont side watersheds and 6 of the 32 indicators for the New York side.

While the indicator table seems to suggest that Quebec and Vermont have more tracking data than New York, the indicators themselves do not allow for easy tracking across jurisdictions. Despite removing program-specific and jurisdiction-specific language where possible from the indicators, the apparent lack of data from New York data may be at least in part an artifact of indicators that are too Vermont-specific. However, Vermont's ongoing TMDL redevelopment has necessitated some increased accountability and a corresponding increase in data collection for the past several years in Vermont which has not been paralleled in New York.

Goal 2: Show areas of strong opportunity for future management by comparing what's been done to the universe of need

One of the clearest results is that there has been a high level of implementation directed toward cleaning up pollution sources on regulated farmsteads (i.e., Medium and Large farmsteads), and on getting wastewater treatment and combined sewer outfalls[1] (CSOs) into compliance – evidence of this can be seen in the low level of reductions still possible to achieve from these sources (Figure 2-3). In many of the watersheds, the ultimate acceptable levels for farmstead BMPs, CSOs and wastewater treatment have already been met. For example, across the Lake Champlain Basin, only six wastewater treatment facilities (WWTFs) have produced 3-year average loads in excess of their current TMDL allocations, equivalent to 5% of all facilities. Likewise, CSOs have been eliminated from most watersheds, and reports by the Vermont Agency of Natural Resources suggest that overflow events at the remaining outfalls are relatively rare in Vermont (Vermont Agency of Natural Resources 2013). Because rules dictating the use of farmstead BMPs for farms regulated under federal programs have existed for many years in the US, very few Medium and Large farms in the Vermont and New York portions of the basin are out of compliance with maintaining these structures at any time. In the Quebec portion of the basin, similar rules for animal farms are more stringent in terms of the size of farms that are closely regulated, and compliance rates are similarly high. Because of the lack of significant management potential, any remaining targets for these policies in these other watersheds would account for only 5% of the possible phosphorus reductions. However, enacting new, more stringent targets could change the degree of potential that exists for some of these policies. For example, lowering the allocation for WWTFs or regulating Small Farm Operations to the

[1] We use "outfall" in reference to the outfall pipe where combined sewer systems are discharged to a stream, and "overflow" to describe events when such a discharge occurs. In abbreviation, "CSO" refers to the outfalls, and "CSO events" refers to overflow events.

same level as their larger counterparts would present some additional opportunity for reductions. As our focus was on existing regulatory programs, estimating the effects of these hypothetical changes was outside the scope of this effort.

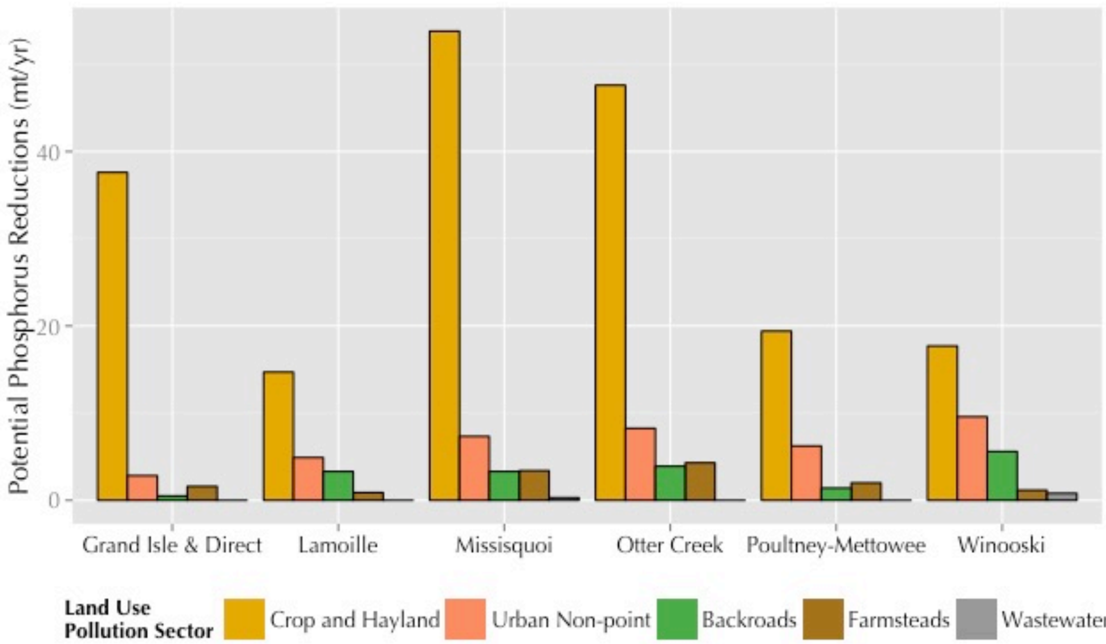

Figure 2-3. Potential reductions by land use pollution sector and tributary in the Vermont and Quebec portions of the Lake Champlain Basin.

On the other hand, management agencies can still work to further encourage better management of agricultural fields and treatment of runoff from impervious surfaces, the two pollution source categories with the largest potential for phosphorus reductions (Figure 2-3). In most watersheds, only a small percent of agricultural fields are managed with cover crops, alternative manure management (e.g., subsurface injection) or reduced tillage (Figure 2-4). Reduced tillage and manure injection are considerably more common in the Missisquoi Basin because of the intensity of those practices in the Quebec portion of the basin. Throughout the Lake Champlain basin, tracking the prevalence of these and other management practices is difficult because farmers often implement them voluntarily and without any compensation. Additionally, it is possible that these practices could be implemented on the same acres, effectively double counting acreages that are under some level of enhanced management.

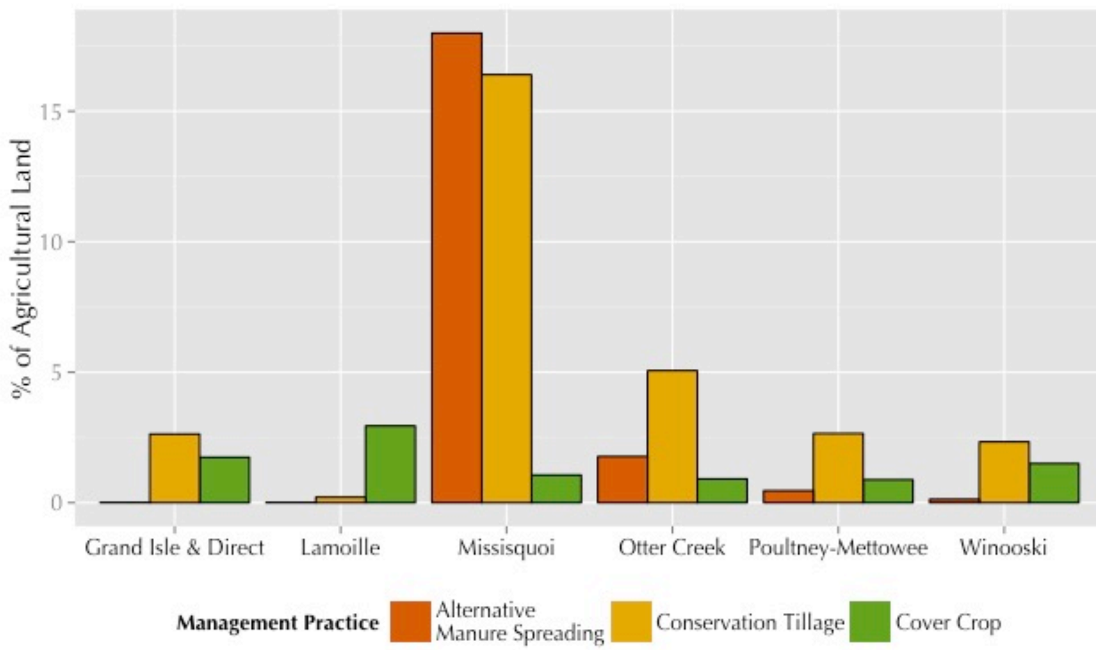

Figure 2-4. Percent of agricultural land under enhanced management in Vermont and Quebec portions of the Lake Champlain Basin.

The data presented in this report only reflect the acres of each practice cost-shared by the Vermont Agency of Agriculture, and likely underestimate the actual rate of use for each management practice. However, we assume that the cost share programs capture most of the acreage, and therefore assume the actual acreage is not more than twice what we report. However, even under-reporting by as much as a factor of 5 would not change the general result that better management of agricultural fields represents the largest opportunity for phosphorus reductions across land use pollution sectors.

The percentage of area of impervious surface under active management is also small (Figure 2-5). Across watersheds, the average proportion of impervious surface under state permit is 5.8%. Without the Grand Isle & Direct drainage and the Winooski basins, both of which have large populated areas (St. Albans and Burlington, respectively), the average percentage of permitted impervious surface is only 2.8%. In basins with urban areas subject to the federal Municipal Separate Storm Sewer System (MS4) rules, which contain regulations for managing stormwater, these estimates could be lower than the actual impervious area under regulation. This discrepancy is a result of the fact that Vermont stormwater permits exist for parcels both inside and outside the MS4 boundaries; therefore, the area of stormwater permits issued by the State and the area of MS4 communities are two separate estimates of the impervious area under management with substantial (but less than perfect) overlap. We have chosen the first, under the assumption that the MS4 designation does not ensure effective stormwater treatment for all impervious parcels.

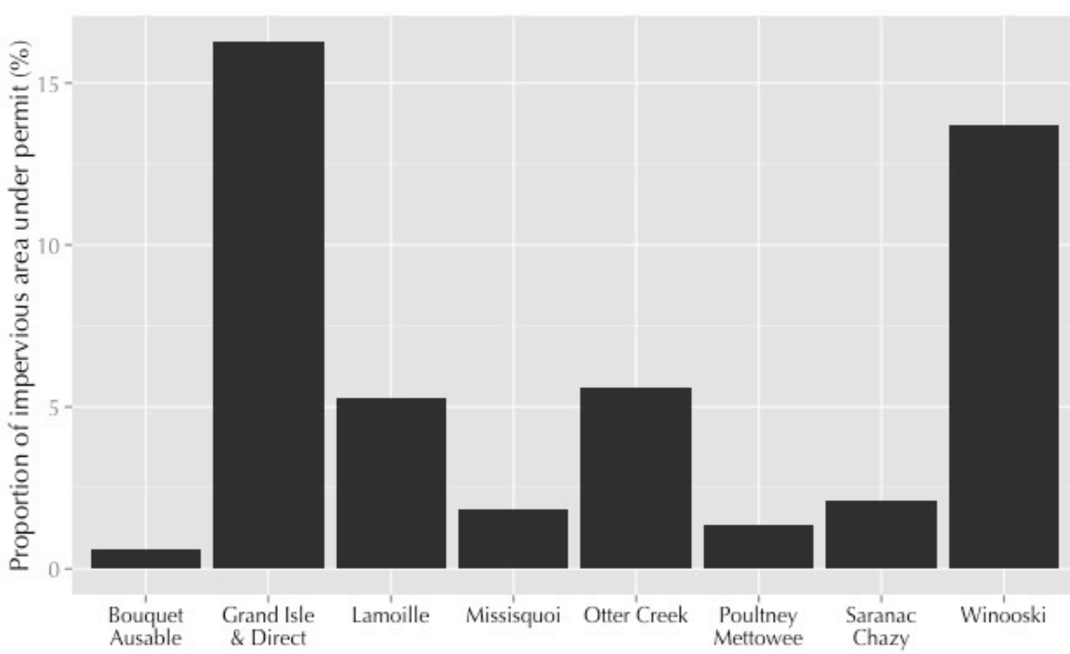

Figure 2-5. Percent of existing impervious area managed under stormwater permits in New York and Vermont portions of the Lake Champlain Basin.

In each jurisdiction, there are different area limits for how large new impervious areas need to be in order to require permitting. This is pertinent in that it appears that much of the impervious area may lie in parcels that are too small to warrant permits. In addition, the proportion of impervious surface that is associated with roads (i.e., impervious surface that is not associated with rooftops, parking lots, etc.) varies from watershed to watershed, which is important for understanding how much of the impervious area is manageable with different practices (Table 2-3).

Table 2-3. Percentage of impervious surface in each watershed associated with roads and railways in the New York and Vermont portions of the Lake Champlain Basin.

Tributary	%
Bouquet-Ausable	57.7%
Grand Isle & Lake Champlain Direct	37.9%
Lamoille	42.7%
Missisquoi	47.5%
Otter Creek	41.1%
Poultney-Mettowee	45.3%
Saranac-Chazy	49.9%
Winooski	38.9%
Mean	*45.1%*

Goal 3a: Provide simple estimates of total reduction potential for each of the major management initiatives tracked in the Indicator table:

The largest potential for land-based phosphorus reductions over the long-term appears to come from managing runoff from crop and hay land and urban non-point sources (i.e., from treating runoff from impervious areas). We estimated that wide-scale management of these two land uses to their ultimate acceptable levels could reduce tributary loads up to 235 metric tons of phosphorus per year – 191 from crop and hayland and 44 from urban areas (Table 2-4). The large difference in reductions possible from these two sources is primarily a result of the modeling data that suggests that the phosphorus loading from agricultural lands consists of more than two-thirds of the loading from non-forested, upland areas. Because the estimates of possible reductions are based on proportions of phosphorus removed through various practices, the higher loading estimates translate to more potential to reduce those loadings. However, both of these estimates – potential reductions from agricultural fields and from urban non-point – are almost certainly optimistic to some degree.

Table 2-4. Potential long-term phosphorus reductions (mt/yr) within each land use pollution sector.

Tributary	Agriculture		Developed Lands		Backroads	Wastewater Treatment	Total
	Fields	Farmsteads[2]	Impervious Area	CSOs			
Grand Isle/Direct	37.6	1.6	2.8	0.03	0.5	0.0	42.53
Missisquoi Bay	53.8	3.4	7.2	0.13	3.3	0.3	68.13
Lamoille	14.7	0.89	4.9	0.0	3.3	0.0	23.79
Winooski	17.7	1.2	9.5	0.08	5.6	0.8	34.88
Otter Creek	47.6	4.3	8.1	0.16	3.9	0.0	64.06
Poultney-Mettowee	19.4[3]	2.0	6.2	0.03	1.4[4]	0.0	29.03
Bouquet-Ausable	--[5]	0.86	2.3	0.0	--	0.0	3.16
Saranac-Chazy	--	1.46	3.0	0.15	--	0.0	4.61
Totals	190.8	15.71	44.0	0.58	18.0	1.1	270.19

[1] Reduction estimates are based on estimates of land use specific phosphorus loadings made by Tetra Tech (2013).
[2] "Farmsteads" refers only to regulated farmsteads - i.e., Medium and Large Farm Operations/Medium and Large CAFOs. Small Farm Operations/Small CAFOs have been excluded from this analysis because of the lack of clarity about how many exist, and because they are currently subject to less stringent regulatory standards.
[3] This value reflects reductions possible from only the Vermont portion of the basin, as no data was available for the extent of these practices on the NY side of the basin.
[4] This value reflects reductions possible from only the Vermont portion of the basin, as no data was available for the phosphorus loading rate from the road network on the NY side of the basin, which is a key element of calculating possible reductions.
[5] Indicates that no data was available to estimate this value.

Firstly, the reductions possible from crop and hayland assume that the three practices we included (i.e., cover crops, reduced tillage, and alternative manure handling practices) could each be implemented on every acre of agricultural field and achieve an additive level of effectiveness. This may not be true. However, there is no good basis for estimating what the combined efficiencies of those three practices might be on a basin-wide scale. In addition, limitations of the soil or terrain might preclude implementation of all three practices simultaneously, though again, no good basis exists for estimating where simultaneous use of the three practices could or could not be used.

Secondly, managing storm water from urban impervious area (IA) often requires retrofitting practices into spaces between existing buildings, parking lots, roads, and other urban infrastructure. Very often it is impossible to retrofit enough practices to treat runoff from all IA, which we've indicated is the ultimate acceptable level for this indicator in the Indicator Table. In addition, not all IA is directly connected to waterways (via stormdrains, or otherwise), and so not every cubic foot of runoff from IA is in equal need of treatment. The IA that is connected to waterways is called effective impervious area (EIA), and though its area can be often substantially less than the total IA, its effect is disproportionately negative. The estimates of possible reductions from urban areas are based on an assumption that all impervious area is capable of receiving treatment, but for the reasons noted here, that is not an entirely realistic goal. Estimates of the extent of total IA that is feasible to treat have come from more densely populated urban areas such as Boston and have indicated that as little as 30% might be a more realistic expectation (Perkins 2013), but given the considerably lower population density of the Lake Champlain Basin's cities, a much higher proportion might be achieved here. More detailed analyses of EIA within each watershed and the potential for retrofits in more densely populated municipalities in Vermont, New York, and Quebec would provide a better estimate of what proportion of IA runoff could receive treatment and in turn what the potential for reductions might be.

In both cases described above, we had the option to develop an arbitrary reduction factor to adjust the reduction estimates according to the uncertainty noted above, or alternatively to acknowledge the uncertainty and leave the estimates in their current form. We opted for the latter path, preferring not to include calculations and adjustments to the data that could not be justified, which can imply a better understanding of what the data represent than the data actually allows (Pilkey-Jarvis and Pilkey 2008). In a similar vein, unstable river channels appear to contribute heavily to phosphorus loading in some watersheds (Tetra Tech 2013a), indicating that a return to geomorphic equilibrium could provide substantial reductions in sediment and nutrient loading. However, we did not estimate these reductions because of the substantial uncertainty about the timescale associated with achieving that equilibrium and the likely scale of the reductions once equilibrium has been achieved.

As discussed in the previous section, ensuring better compliance with current WWTF and CSO targets and addressing remaining farmstead sources with currently-existing regulations hold far less potential for achieving large-scale reductions, in part because a considerable amount of effort has already been exerted to alleviate these sources. Though they both fall under the category of non-point sources, their highly visible nature has made them prime targets for reducing pollution. However, it's apparent that those sources no longer represent serious potential for attaining the level of non-point source phosphorus reductions that are needed to achieve the new Lake Champlain TMDL.

In addition to clear results indicating which pollution sectors hold the most and least promise for achieving large scale phosphorus reductions, the data also make it clear that not all watersheds hold the same potential for reductions – that there is substantial geographic variation. To use agricultural fields as an example, widely implementing the three practices of interest in the Missisquoi, Grand Isle/Direct, and Otter Creek watersheds could lead to reductions of up to 125 mt/yr, which is over 70% of the total potential for all watersheds from the agricultural sector. Similarly, treating runoff from impervious areas in the Missisquoi, Winooski, and Otter Creek watersheds would address more than 50% of the potential reductions from that source category. This geographic variability represents an opportunity to apply the critical source area concept, which suggests that a small proportion of the landscape contributes disproportionately to water quality impairments. Local studies have shown that this is indeed the case, at least in the Missisquoi basin (Ghebremichael and Watzin 2010, Winchell et al. 2011). Addressing specific pollution sectors in watersheds where they are most significant provides an opportunity to make significant progress toward large reductions faster by targeting efforts into smaller geographic areas to overcome thresholds in ecosystem response (Scheffer and Carpenter 2003).

Goal 3b: Provide simple estimates of cost-effectiveness for each of the major management initiatives tracked in the Indicator table:

Total cost and cost-effectiveness also vary greatly by pollution sector and by watershed. Figure 2-6 shows the total cost to achieve the reductions noted in table 3. Not surprisingly, larger watersheds will incur higher costs for addressing their larger total phosphorus loads (Table 2-5). However, variation in the relative proportions of each land use within watersheds introduces some differences in the costs between watersheds apart from their size and indicates that differences in cost-effectiveness between sectors is an important consideration.

In terms of total cost, addressing stormwater in urban settings is extremely expensive (and highly variable) because of the high cost of retrofitting treatment structures into small spaces within existing infrastructure. The cost data shown here for each set of practices reflect both the initial investments – base construction costs and design and engineering costs – and estimates of annual operation and maintenance costs. In the current form of the indicator table we also calculated cost for initial investments only, since there is enormous disparity between upfront costs associated with treating urban non-point, where less than one percent of the total long-term cost is associated with annual maintenance, and the use of cropland BMPs, which present a regular annual expense. Even when extrapolating these costs out over twenty years, Figure 2-6 and Table 2-5 show clearly that the total cost of treating urban stormwater far exceeds the cost for managing agricultural fields, farmsteads, and backroads. In the Missisquoi watershed, the difference between the cost to manage agricultural fields and urban stormwater is small. However, in every other watershed, the difference is a factor of 2 to 10. The much larger potential for management of impervious area and of agricultural fields is the main driver of the very large cost to achieve high levels of management in these two sectors.

Table 2-5. Estimated 20-year costs ($ millions) to achieve phosphorus reductions identified in Table 2-4.

Tributary	Agriculture		Developed Lands		Backroads	Total
	Fields	Farmsteads[1]	Impervious Area	CSOs		
Grand Isle/Direct	37.12	1.68	88.38	25.05	1.26	153.49
Missisquoi Bay	137.88	4.57	182.21	7.60	35.67[2]	367.93
Lamoille	27.41	0.94	237.10	0.0	2.72	268.17
Winooski	38.30	1.93	484.40	49.63	2.99	577.25
Otter Creek	87.23	3.76	310.72	17.45	2.33	421.49
Poultney-Mettowee	64.15[3]	1.19	440.66	18.57	0.83[2]	524.65
Bouquet-Ausable	--[4]	1.52	291.05	0.0	--	292.57
Saranac-Chazy	--	1.16	343.21	49.07	--	393.44
Totals	392.17	16.75	2377.73	167.37	45.8	2999.82

[1] "Farmsteads" refers only to regulated farmsteads - i.e., Medium and Large Farm Operations/Medium and Large CAFOs. Small Farm Operations/Small CAFOs have been excluded from this analysis because of the lack of clarity about how many exist, and because they are currently subject to less stringent regulatory standards.

[2] This value reflects costs for only the Vermont portion of the basin, as no data were available for the extent of these practices on the NY side of the basin.

[3] This value reflects costs from only the Vermont portion of the basin, as no data on road BMPs were available for the NY side of the basin.

[4] Indicates that no data were available to estimate this value.

Figure 2-7 shows that when effectiveness is taken into account, however, the picture can change slightly. While controlling urban pollution is still very costly[2], treating farmstead

2 Cost-effectiveness estimates for CSO elimination have been excluded from the urban non-point category in Figure 2-7. At an average of $35,000 per kilogram of phosphorus, those data obscure differences between the

runoff, with its relatively lower effectiveness, can become more similar in terms of cost per kilogram of phosphorus, particularly in watersheds where there are higher numbers of farms still requiring structural BMPs. The effectiveness of treating stormwater ranges from roughly $1400 per kilogram of phosphorus in watersheds where the impervious area is small, to roughly $3000 per kilogram of phosphorus in watersheds with high levels of impervious surface. Implementing field practices in agricultural cropland and making wide use of backroad BMPs are orders of magnitude more efficient than their counterparts.

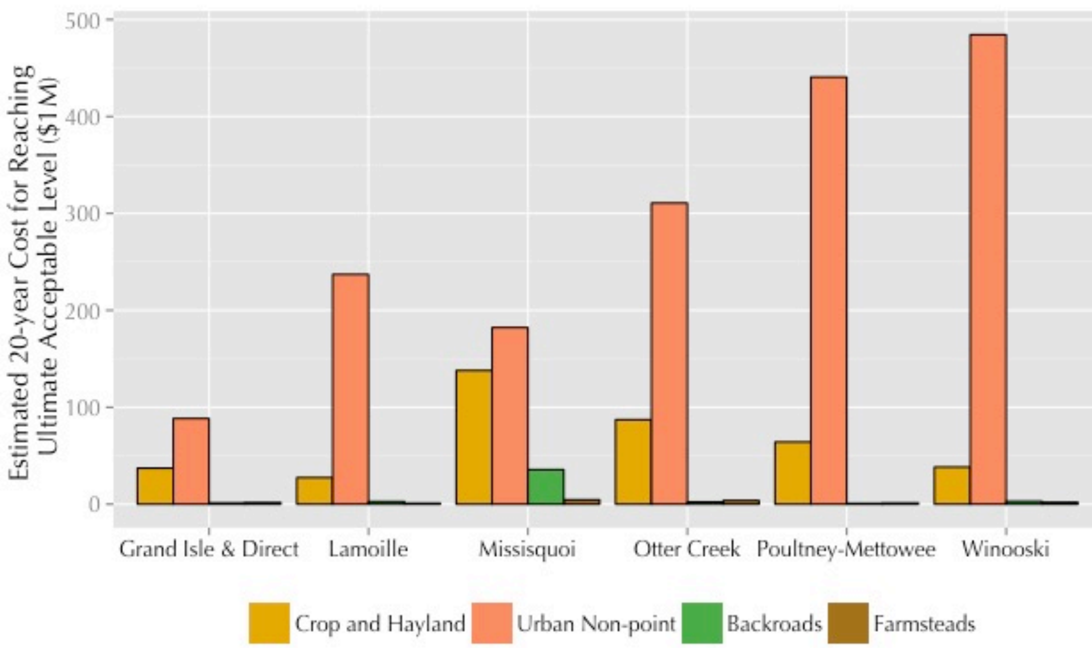

Figure 2-6. Total cost to achieve ultimate acceptable levels set in the Indicator table by land use sector.

Goal 4: Identify important knowledge gaps in our understanding of what management has occurred or in the effects of that management:

One key element of an adaptive approach to any sort of resource management, including improving water quality, is to clearly articulate any gaps in knowledge or major sources of uncertainty. This process of articulating what is unknown and how those

other categories and distort the cost-effectiveness of treating stormwater, which represents the bulk of the real phosphorus reduction opportunity.

uncertainties may impact current decision-making can often point the way toward research efforts that will truly improve the long-term effectiveness of management decisions. During the course of this project, we uncovered three major categories of gaps in the collective knowledge about the management of Lake Champlain, each with different implications for the results discussed above.

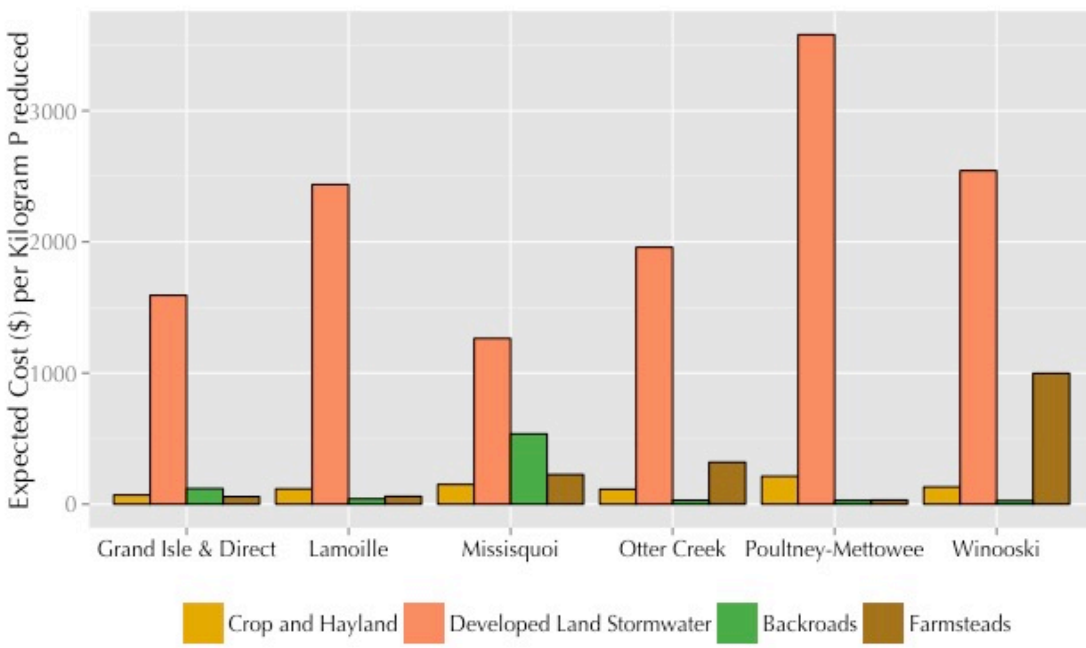

Figure 2-7. Average cost-effectiveness across practices within land use sectors. Urban non-point excludes CSO elimination due to its very high cost.

The first major category of uncertainty is the role of variability in several of the factors key to the Indicator table estimates of reductions and cost-effectiveness. One key part of the method to calculate potential reductions used values of treatment efficiencies (called reduction rates) for each practice. The estimates we used reflected average values seen in studies in the scientific literature. However, unlike many of the other factors used in the calculations, treatment efficiencies were applied identically throughout the Lake Champlain basin when in fact there is likely to be a large amount of variation in the effectiveness of each practice at various implementation sites within and among sub-basins. Though the single estimate may be a reflection of the average performance across a wide range of conditions, the true performance within a single watershed – even a large one – may depart from that average enough to lead to different estimates of potential reductions.

Box 3. Variability & Uncertainty

In structured decision making and adaptive management contexts, the terms variability and uncertainty refer to different concepts, and their use in this report reflects that distinction.

Variability refers to the property of predictable variation around an expected value. The practice of statistics often expresses this predictability as the standard deviation of the mean, and reliably characterizing the standard deviation requires a number of samples of the quantity in question. While more data can lead to more precise estimates of variability, the inherent variability of a population cannot be reduced.

Uncertainty, on the other hand, refers to the situation where there is no expected value for the quantity in question. Uncertainty can occur as a result of a lack of applicable data, or as a result of major disagreements between existing datasets. When uncertainty stands in the way of good decision-making, expert elicitation methods can be used to generate defensible expert opinion (Morgan and Henrion 1990). In contrast to variability, new data can and do reduce uncertainty.

This same variability also can play a significant role in the estimates of cost to implement management policies. The Center for Watershed Protection, a non-profit organization focused on watershed management issues, noted in their Urban Subwatershed Restoration Manuals that the cost to implement individual stormwater retrofits can vary over a factor of three to ten, depending on the type of practice in question (Schueler et al. 2007). This variability in cost is driven by several considerations that also vary geographically and over time, including the spatial arrangement of existing infrastructure, the complexity of the design process, the need for permitting, the cost of land, and local labor rates. All of this variability is important to consider in the cost estimates that we use for the agricultural practices, urban stormwater practices, backroad maintenance practices.

Unfortunately, we were unable to characterize this variability sufficiently well to use that information in calculating potential reductions or cost-effectiveness estimates for this iteration of the Indicator table. However, in future revisions of the table, as new data are incorporated, estimates of the variability for both reduction rates and practice costs will begin to emerge and can be incorporated into new calculations. While new data can help quantify and characterize this sort of variability, it is important to remember that the variability cannot be reduced (see box 3). Characterizing and accounting for the variability in treatment efficiencies, for example, serves mainly to understand the range of potential reductions that might be expected given the information at hand, and therefore help to reduce the magnitude of any surprises when the average reductions and average costs don't apply to all watersheds equally.

The second major source of uncertainty that we uncovered is often referred to as parameter uncertainty. This describes the situation where no good average estimate exists, and we are forced to pick one that we know has limitations. This is in contrast to data variability, where we were able to find and use a good average estimate for something of interest in a calculation but unable to know how much that estimate might vary from location to location. The clearest example of this situation in the Indicator table is in estimates for the current extent of management practices. Many of these estimates have some limitations that we were unable to avoid, often due to gaps in how these practices are tracked within and across jurisdictions. Specifically, the rate of implementation of cover crops is subject to a large amount of uncertainty. The data that we present here comes from the Vermont Agency of Agriculture (VT AAFM), which provides payment to farmers who implement cover crops on their fields. However, farmers can also enroll in cost-share programs administered by the USDA Natural Resources Conservation Service (NRCS), which may provide additional cost share to those same VTAAFM-funded fields, or provide payment for different fields. Many farmers also implement cover crops without receiving compensation, either because they exceed the acreage caps set by VT AAFM and NRCS, or because they believe that cover crops are an economically or environmentally beneficial practice for their farm. Therefore, the VT AAFM data do not account for all cover cropping that occurs in Vermont and the degree of overlap with NRCS cover cropping programs and the extent of voluntary cover cropping is unknown. This category of uncertainty also applies to other management practices including structural farmsteads BMPs, and stormwater management practices.

The third gap in our knowledge concerns the environmental effects of management actions, particularly in large watersheds. Many of the reduction estimates presented here used reduction rates that were determined in small-scale site-level studies or, in some cases, small watershed studies. However, larger-scale studies have shown that when these practices are implemented widely across a watershed that the reduction rates are often far less than they appear at small scales, i.e., watershed-level reductions are not the sum of field-level reductions. For example, Meals (1996) found that when measured at the field-scale, several practices, including the installation of vegetated filter strips and the elimination of winter manure spreading, led to very high phosphorus reduction rates, but that at the watershed scale, phosphorus load reductions were not significant over an 11-year monitoring period. Similarly, Davie and Lant (1994) predicted reduction of stream sediment loads in two large watersheds of 24% and 37% after widespread enrollment of agricultural land into the USDA Conservation Reserve Program (CRP). However, in the three years following the CRP enrollment, sediment exports were reduced just 0.0125% and 0.265% in each watershed. Both of these studies indicate that in-stream (and potentially other) processes play an important role in dampening the effects of upstream management and contribute to long lag times in realizing these effects downstream. This hypothesis is supported by numerous studies of lag times and the driving role of climate, flow, and in-stream re-suspension of sediments in generating nutrient loads (Richards et al. 2009, Meals et al. 2010, Niemitz et al. 2013). Understanding how in-stream processes mitigate the effect that management practices have on phosphorus loads would enable us to generate more realistic predictions of how long it will take to achieve large-scale phosphorus reductions and how large those reductions could be.

Conclusions & Applications:

Using the Indicator Table for decision making

Our primary intention is for these data to inform future management decisions for Lake Champlain. Therefore, the rest of this discussion focuses on potential uses for the data moving forward, and not on a discussion of the effectiveness of past management policies.

The measures of the universe of need, expected phosphorus reductions, and expected cost to achieve those reductions for each indicator are all meant to be inputs to a rigorous and analysis-focused decision-making process focused on the best way to achieve wide-scale reductions in tributary phosphorus loads. These sorts of decision-making processes use data to enable decision-makers to weigh several alternative management strategies against each other according to their likely outcomes, and to assess the tradeoffs that would be required by selecting any combination of the alternatives (Keeney 1982, Gregory and Keeney 2002). There are two natural forums for this sort of process in the context of managing phosphorus in Lake Champlain. The first is in the redevelopment of commitments in *Opportunities for Action*, where the LCBP and its partners commit to accomplishing a suite of high-priority phosphorus management targets over the following five-to-seven years. The second is in the development and refinement of Tactical Basin Plans in Vermont, which will follow the approval of Vermont's revised Lake Champlain Phosphorus TMDL in 2014, and in similar watershed planning efforts in New York and Quebec.

One of the primary aims of revising the commitments in OFA is for LCBP's partner agencies to commit to a suite of management policies that together constitute a coordinated and coherent management strategy that reflects recent progress, current management priorities, and best professional judgment of the most effective policies and practices. A common sense evaluation of the data we present would indicate that OFA commitments focusing on implementing phosphorus conservation practices on crop and hayland by the LCBP and its partners in the Lake Champlain basin would provide the best use of time and financial resources to reduce tributary phosphorus loads, because that pollution sector presents the most cost-effective and wide-reaching phosphorus reductions. However, this common sense approach assumes that 1) resources are limited to the extent that full management of every pollution sector is not possible, 2) the only two important factors for setting strategic management priorities are total reduction potential and total cost, and that 3) implementation is the best use of available funds. While the first of these assumptions is likely to be true, the other two are not.

In addition to total reduction potential and total cost of implementation, each of the LCBP partners would likely identify several other criteria that are more or less important or desirable for guiding management priorities (Gregory 2013). Some of these criteria might include the timeliness of management effects (e.g., policies with long-term effects vs. short-term effects), the ability to leverage existing legislation to encourage further reductions, equity in cost and benefit[3] between geographic sections of the lake (e.g., jurisdictions, north vs. south lake), equity in cost and benefit between different pollution sectors (urban vs. agricultural), or equitable distribution of responsibility between public and private entities. A thoughtful process of eliciting and weighing these and other criteria and understanding how new OFA commitments perform in relation to these criteria is clearly an important step toward making commitments to more efficient and effective phosphorus management policies. Multiple criteria decision analysis (MCDA) methods are designed to do exactly this and are a core element in SDM practice. The data we report here are intended to be an input to this process, as some of the conclusions lend themselves directly to inclusion in an SDM process rooted in MCDA.

The third assumption listed above is consistent with the higher value that organizations and the public generally place on implementation of management actions relative to other activities associated with environmental management, such as monitoring or research (Allan and Curtis 2005). However, monitoring and research have clear value for understanding variability in BMP efficiency and for reducing uncertainty, and many studies have documented that investments in such activities pay for themselves over time in the context of resource management (Borisova et al. 2005, Kangas et al. 2010, Williams et al. 2011). There are two important ways that research and monitoring can improve the decision-making process and outcomes. Firstly, variability in any estimate, such as the phosphorus reduction rate of a particular management practice in a watershed or the cost to implement that same practice in a new watershed, often interacts with the variability of other estimates in somewhat unpredictable ways. These interactions can produce surprising and

[3] The term "cost" is meant in this context to include the social cost of being recognized as responsible for the lake's impaired status in addition to realized financial burden. "Benefit" is meant to recognize that funding streams are often concomitant with the responsibility for environmental clean-up activities.

unforeseeable results, particularly if the range of variability for each estimate is not well understood. For example, while we report the average reduction rate for cover cropping across the Champlain basin, the actual reduction rate in any tributary may be higher or lower than the average. Similarly, the actual cost to implement cover crops in that same tributary may also depart from the average, and the interaction of these sources of variability may mean that the cost-effectiveness of cover cropping on a local scale may be much higher or much lower than expected based on the average estimates. Understanding the extent of variability for each estimate and how they might interact can help decision-makers generate a more realistic range of outcomes, reducing the likelihood of a result that deviates widely from expectations.

While we did not perform any Value of Information analyses, our data do point to several examples of variability that are good candidates for more explicit characterization, including reduction rates for the most common management practices, and the costs to implement those practices. These figures are clearly key to the estimates of potential reductions for each watershed and the estimates of cost effectiveness, both of which are critical pieces of information for good decision making. The data also indicate that some values, while subject to high variability, are probably not worth further characterization because the result of better understanding would not inform future decisions differently. For example, phosphorus loading from combined sewer overflow events is subject to very high variability because of the sporadic timing, unpredictable water quality of the effluent, and the wide range of volumes and intensities that can occur in these events. However, the total loading from CSOs accounts for 1.3% of the urban phosphorus load basin wide, and further understanding of CSO events is unlikely to move their management from its current low-priority status (in terms of phosphorus) to a high priority status in the context of nutrient management (though this may be a different priority in the context of toxin pollution reduction).

The second benefit of research and monitoring is that these activities, when designed to target specific uncertainties, can shed light on the complex relationships between policies and their environmental effects. Uncovering these relationships can reduce the length of time that an ineffective policy is relied upon before being changed, or, more ideally, can reduce the likelihood of implementing ineffective policies in the first place (Morgan and Henrion 1990, Lempert et al. 2003, NRC 2007). In addition, gaining this understanding can generate more broadly applicable knowledge about how ecosystems respond to management and about what forms and targets of management are most effective. Developing this deeper ecological understanding is the one of the key intentions of the adaptive management approach (Walters 1986, Gunderson 2001), and can be extremely useful for addressing other similar management problems.

A second application of these data is to assist the State of Vermont in the process of developing Tactical Basin Plans to achieve the loading targets set in Vermont's Total Maximum Daily Load (TMDL) for phosphorus, which is currently under revision by the EPA. As part of the TMDL revision effort, the EPA developed a scenario tool to assist the State of Vermont in implementing a plan to achieve the TMDL reduction recommendations. These data could be of great use for developing scenarios in each major basin to achieve the loading targets set in the TMDL. Because the Indicator Table makes slightly different assumptions and uses slightly different data sources than the EPA scenario tool, the different estimates of potential reductions they produce can provide additional information that each tool could not provide independently. These small differences between the tools' estimates can lend some insight into the range of uncertainty in the calculations and the effects of the assumptions of each method. These insights in turn help to provide greater confidence when the results are similar, and can point to key assumptions in need of further investigation when the results are not (Arabi et al. 2012). In both situations, the average model predictions are generally more accurate than either individual model, which helps to produce more realistic expectations on the part of the decision-makers and the public (Osmond et al. 2012). The practice of using multiple models in this way for complex problems has gained wide use in recent years because of the increased understanding that decision makers get from seeing multiple solutions to a problem (Lempert et al. 2003, NRC 2007). In the context of adaptive management, the use of multiple models has become commonplace not only because of the ease with which the data can be integrated in statistically defensible ways, but also because the difference in performance between models can indicate the level of overall uncertainty in the system (Martin et al. 2011), and because over time model performance can be compared to monitoring results to provide increased confidence in the predictions from a subset of the models (Johnson and Williams 1999).

Maintaining and updating the Indicator Table

Adaptive management requires a continual process of management decisions, monitoring their outcomes, and then using new monitoring information to inform the next round of decisions. In order to be effective over the long term, all parts of the Indicator Table, including the data elements and the indicators themselves, should be revisited at regular intervals as better information becomes available. In particular, Current State information (for both the Implementation and Ecosystem State indicators) should continue to be updated with new monitoring data, and practice reduction rates and unit cost estimates should be updated as studies within the Lake Champlain basin develop more locally-relevant data.

One of the purposes of adaptive management frameworks is to provide an explicit and regular opportunity for new monitoring data to be used to revisit and update management objectives as more is learned about the feasibility of attaining specific objectives. Over time, as the quality of the data increases, new analyses of those data should in turn inform the revision of management objectives held by the LCBP and its partners. New indicators will be added to the Indicator Table as new management initiatives are developed, and existing indicators will be eliminated as the initiatives they represent are de-emphasized. Because the Indicator Table is intended as a decision-aid for the LCBP's management strategy, the Indicator Table should be revised in preparation for OFA updates.

References

Allan, C., and A. Curtis. 2005. Nipped in the bud: Why regional scale adaptive management is not blooming. Environmental Management **36**:414-425.

Allen, C. R., J. J. Fontaine, K. L. Pope, and A. S. Garmestani. 2011. Adaptive management for a turbulent future. Journal of Environmental Management **92**:1339-1345.

Arabi, M., D. W. Meals, and D. L. Hoag. 2012. Watershed Modeling: National Institute of Food and Agriculture - Conservation Effects Assessment Project.*in* D. L. Osmond, D. W. Meals, D. L. Hoag, and M. Arabi, editors. How to Build Better Agricultural Conservation Programs to Protect Water Quality: The National Institute of Food and Agriculture – Conservation Effects Assessment Project Experience. Soil and Water Conservation Society, Ankeny, IA.

Borisova, T., J. Shortle, R. D. Horan, and D. Abler. 2005. Value of information for water quality management. Water Resources Research **41**.

Caulkins, J. P. 2002. Using models that incorporate uncertainty. Journal of Policy Analysis and Management **21**:486-491.

Davie, D. K., and C. L. Lant. 1994. The effect of CRP enrollement on sediment loads in two southern Illinois streams. Journal of Soil and Water Conservation **49**:407-412.

EPA. 2000. Evaluation Guidelines for Ecological Indicators. US EPA Office of Research and Development, Washington DC.

Garton, J. 2013. Personal communication, Nov. 12, 2013.

Ghebremichael, L. T., and M. C. Watzin. 2010. An Environmental Accounting System to Track Nonpoint Source Phosphorus Pollution in the Lake Champlain Basin, Technical Report No. 60. Lake Champlain Basin Program, Grand Isle, VT.

Gitau, M. W., T. L. Veith, W. J. Gburek, and A. R. Jarrett. 2006. Watershed level best management practice selection and placement in the town brook watershed, New York. Journal of the American Water Resources Association **42**:1565-1581.

Gregory, R. 2013. Decision-Aiding for the Lake Champlain Basin Program: Report of a Two-Day Program Overview, February 26/27, 2013. Prepared for the Lake Champlain Basin Program.

Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. Structured Decision Making: A Practical Guide to Environmental Management Choices. Wiley-Blackwell.

Gregory, R. S., and R. L. Keeney. 2002. Making smarter environmental management decisions. Journal of the American Water Resources Association **38**:1601-1612.

Gunderson, L. H. 2001. Managing surprising ecosystems in southern Florida. Ecological Economics **37**:371-378.

Hirsch, R. M., D. L. Moyer, and S. A. Archfield. 2010. Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay river inputs. JAWRA Journal of the American Water Resources Association **46**:857-880.

Johnson, F., and K. Williams. 1999. Protocol and practice in the adaptive management of waterfowl harvests. Conservation Ecology **3**:article 8. [online] URL: http://www.ecologyandsociety.org/vol3/iss1/art8/.

Johnson, F. A., W. L. Kendall, and J. A. Dubovsky. 2002. Conditions and limitations on learning in the adaptive management of mallard harvests. Wildlife Society Bulletin **30**:176-185.

Kangas, A. S., P. Horne, and P. Leskinen. 2010. Measuring the Value of Information in Multicriteria Decisionmaking. Forest Science **56**:558-566.

Keeney, R. L. 1982. Decision-analysis - an overview. Operations Research **30**:803-838.

Lake Champlain Study. 1979. Shaping the Future of Lake Champlain. Final Report. Burlington, VT.

Lempert, R. J., S. W. Popper, and S. C. Bankes. 2003. Shaping the next one hundred years: New methods for quantitative, long-term policy analysis. RAND-Pardee Center, Santa Monica, CA.

Martin, J., P. L. Fackler, J. D. Nichols, M. C. Runge, C. L. McIntyre, B. L. Lubow, M. C. McCluskie, and J. A. Schmutz. 2011. An Adaptive-Management Framework for Optimal Control of Hiking Near Golden Eagle Nests in Denali National Park. Conservation Biology **25**:316-323.

Meals, D. W. 1996. Watershed-scale response to agricultural diffuse pollution control programs in Vermont, USA. Water Science and Technology **33**:197-204.

Meals, D. W., S. A. Dressing, and T. E. Davenport. 2010. Lag Time in Water Quality Response to Best Management Practices: A Review. Journal of Environmental Quality **39**:85-96.

Medalie, L., R. M. Hirsch, and S. A. Archfield. 2012. Use of flow-normalization to evaluate nutrient concentration and flux changes in Lake Champlain tributaries, 1990–2009. Journal of Great Lakes Research.

Morgan, M. G., and M. Henrion. 1990. Uncertainty. Cambridge University Press, Cambridge.

Niemitz, J., C. Haynes, and G. Lasher. 2013. Legacy sediments and historic land use: Chemostratigraphic evidence for excess nutrient and heavy metal sources and remobilization. Geology **41**:47-50.

NRC. 2007. Models in Environmental Regulatory Decision Making. National Academies Press, Washington, D.C.

O'Neil-Dunne, J. 2013. Mapping Impervious Surfaces in the Lake Champlain Basin. Technical Report No. 76. Lake Champlain Basin Program, Grand Isle, VT.

Osmond, D. L., D. W. Meals, D. L. Hoag, M. Arabi, A. E. Luloff, G. D. Jennings, M. L. McFarland, J. Spooner, A. N. Sharpley, and D. E. Line. 2012. Synthesizing the Experience of the 13 National Institute of Food and Agriculture - Conservation Effects Assessment Project Watershed Studies: Present and Future.*in* D. L. Osmond, D. W. Meals, D. L. Hoag, and M. Arabi, editors. How to Build Better Agricultural Conservation Programs to Protect Water Quality: The National Institute of Food and Agriculture - Conservation Effects Assessment Project Experience. Soil and Water Conservation Society, Ankeny, IA.

Perkins, E. 2013. Personal communication, Nov. 6, 2013.

Pilkey-Jarvis, L., and O. H. Pilkey. 2008. Useless Arithmetic: Ten Points to Ponder When Using Mathematical Models in Environmental Decision Making. Public Administration Review **68**:470-479.

Pulwarty, R. S., and T. S. Melis. 2001. Climate extremes and adaptive management on the Colorado River: Lessons from the 1997-1998 ENSO event. Journal of Environmental Management **63**:307-324.

Reckhow, K. 2003. On the need for uncertainty assessment in TMDL modeling and implementation. Journal of Water Resources Planning and Management-Asce **129**:245-246.

Richards, R. P., D. B. Baker, and J. P. Crumrine. 2009. Improved water quality in Ohio tributaries to Lake Erie: A consequence of conservation practices. Journal of Soil and Water Conservation **64**:200-211.
Scheffer, M., and S. R. Carpenter. 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends in Ecology & Evolution **18**:648-656.
Schueler, T., D. Hirschman, M. Novotney, and J. Zielinski. 2007. Urban Stormwater Retrofit Practices. Ellicott City MD.
Shirmohammadi, A., I. Chaubey, R. D. Harmel, D. D. Bosch, R. Munoz-Carpena, C. Dharmasri, A. Sexton, M. Arabi, M. L. Wolfe, J. Frankenberger, C. Graff, and T. M. Sohrabi. 2006. Uncertainty in TMDL models. Transactions of the Asabe **49**:1033-1049.
Stankey, G. H., R. N. Clark, and B. T. Bormann. 2005. Adaptive Management of Natural Resources: Theory, Concepts, and Management Institutions. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station,, Portland, OR.
Stickney, M., C. Hickey, and R. Hoerr. 2001. Lake Champlain Basin Program: Working together today for tomorrow. Lakes & Reservoirs: Research & Management **6**:217-223.
Tetra Tech. 2013. Lake Champlain Basin SWAT Model Configuration, Calibration and Validation. Prepared for EPA Region 1 - New England, Boston MA.
Vermont Agency of Natural Resources. 2013. Reported sewage overflows since 2007. URL: http://www.watershedmanagement.vt.gov/ww/om/Overflows/ReportedSewageOverflowsSince2007.pdf.
Walters, C. J. 1986. Adaptive Management of Renewable Resources. Macmillan Publishing Company, New York, NY.
Watzin, M. C., R. L. Smyth, E. A. Cassell, W. C. Hession, R. E. Manning, and D. Wang. 2005. Ecosystem Indicators and an Environmental Score Card for the Lake Champlain Basin Program. Technical Report No. 46. Lake Champlain Basin Program, Grand Isle, VT.
Williams, B. K., and E. D. Brown. 2012. Adaptive Management: The U.S. Department of the Interior Applications Guide. Department of the Interior Adaptive Management Working Group, Washington DC.
Williams, B. K., M. J. Eaton, and D. R. Breininger. 2011. Adaptive resource management and the value of information. Ecological Modelling **222**:3429-3436.
Winchell, M., D. W. Meals, S. Folle, J. Moore, D. C. Braun, C. DeLeo, and K. Budreski. 2011. Identification of Critical Source Areas of Phosphorus Within the Vermont Sector of the Missisquoi Bay Basin. Technical Report No. 63b. Lake Champlain Basin Program, Grand Isle, VT.
Wolfslehner, B., and H. Vacik. 2011. Mapping indicator models: From intuitive problem structuring to quantified decision-making in sustainable forest management. Ecological Indicators **11**:274-283.

CHAPTER 3. IMPROVING EXPORT COEFFICIENT MODELS THROUGH A BAYESIAN HIERARCHICAL APPROACH

Introduction

Models have become an essential tool for helping to effectively manage non-point source nutrient pollution. In many regions of the US, non-point source (NPS) nitrogen and phosphorus loss from urban and agricultural land uses is a primary cause of accelerated eutrophication, which in turn presents impairments to human uses as well as aquatic health (Carpenter et al. 1998). A critical first step in the effective management of NPS pollution is the ability to estimate the relative magnitude of its sources so that management actions can be targeted appropriately.

Export Coefficient Models (ECMs) have gained wide use for estimating the nutrient export rates for particular land uses – and therefore their relative contribution to loading – because of their limited data requirements and relatively simple implementation, especially compared to more complex process-focused models (Reckhow et al. 1980, Johnes 1996, Borah et al. 2006, Shirmohammadi et al. 2006). Based on a multiple linear regression approach, ECMs for NPS nutrient pollution assume that observed loads can be estimated as a function of land use (Carpenter et al. 1998, Allan 2004). The regression coefficients from these models are interpretable as a unit area rate of export, and are expressed as mass exported per area per time (e.g., kg/ha/yr).

However, as they are traditionally implemented, many ECMs provide only crude estimates of the export rates because they are unable to account for uncertainty in land use classification, nutrient load estimation, watershed-to-watershed differences in topography, soils, and other factors that influence nutrient loading, or annual variability in discharge.

Additionally, input data is generally not normalized for watershed size and the unequal variances in loads that result.

Multiple regression assumes that the predictors are measured without error (Riggs et al. 1978). In fact, this is rarely true for land use classifications. Land use maps can and do have large classification error rates, only some of which of is systematic and can therefore be removed. The uncertainty that remains after this bias-adjustment is particularly problematic for land use classes that have similar spectral signatures (e.g., land uses dominated by grass, such as herbaceous wetland and permanent hayland) (Wickham et al. 2010, Wickham et al. 2013). Where large error does exist in the areal estimates of land uses, estimates of the coefficients will be biased toward 0 (Riggs et al. 1978), limiting their usefulness in estimation of watershed loading and the effects of management actions.

Watershed-to-watershed differences in land use history, topography, or soil types can lead to large differences in average loading rates, but these differences are often lost through averaging across watersheds, even though they are important in a management context.

In many cases, stream monitoring datasets include watersheds that are of significantly different sizes. Classical multiple regression assumes constant variance across the range of the data, and violations of this assumption can lead to poor predictions. It is often possible to satisfy this assumption by averaging over years or across watersheds, but this technique loses important information.

A better method of solving for land use specific nutrient generation rates should 1) include error in the predictors, 2) consider watershed-specific differences that may affect loading rates, 3) include the hydrologic variation within watersheds in the estimates of the nutrient generation rates, and 4) incorporate data from parallel and previous studies as well as theoretical constraints on the values for the coefficients.

Bayesian approaches to ECMs have been implemented to demonstrate the value of integrating different data sources and to quantify general uncertainty in the values of the coefficients for a variety of nutrients (Zobrist and Reichert 2006, Broad and Corkrey 2011, Liu and Lu 2013).

Bayesian hierarchical regression models allow the prediction of individual values based on both individual- and group-level predictors (Gelman and Hill 2009). They have been successfully employed in other water quality management contexts, (Cha et al. 2010, Gudimov et al. 2012), but to our knowledge, this approach has not been applied to the estimation of land use specific phosphorus generation rates.

The goal of this study was to develop a Bayesian hierarchical model that:

a) incorporates and propagates errors in the land use classification,
b) groups watersheds that share similar soil, topographic, or historic land use conditions, and models the effect of these group memberships with varying intercepts,
c) models variance at the group level (rather than assuming constant variance across groups, and
d) makes use of theoretical considerations and experimental data through the use of prior distributions on the export coefficients.

We applied our model to estimate total phosphorus loading rates for seven land use classes across seventeen watersheds of widely varying sizes, land use distributions, and land use history in the Lake Champlain Basin in the northeastern US over the period 2006-2013 to demonstrate a process for estimation of these generation rates that provides more useful information for understanding nutrient loading.

Methods

Study watersheds description:

Lake Champlain is a large lake (1127 km^2) situated between the U.S. states of Vermont and New York, and the Canadian province of Quebec (Figure 3-1, inset). Over the past several decades, excess non-point source phosphorus loading has contributed to an increase in cyanobacteria blooms during the summer months in some areas of the lake (LCBP 2012). As a result, the control of non-point phosphorus is of primary concern to the management community in the Lake Champlain basin.

Seventeen of the tributary watersheds to Lake Champlain (shown in figure 1) with areas ranging from 115 km^2 to 2737 km^2 were selected for inclusion in this study because of their continuous phosphorus and discharge sampling records from 1991 through to the present. Land use in each of the watersheds is mixed. Some areas in the eastern and southern parts of the basin are used intensively for agricultural purposes, while much of the western portion is protected. Urban areas (with a few exceptions) are highly localized in the lower-elevation areas of the watersheds, close to the lake's shoreline.

We delineated contributing areas upstream of each water quality monitoring location based on a 10m digital elevation model (DEM) of the Lake Champlain Basin (VCGI 2006) and using the suite of hydrologic modeling tools in the Hydrology toolset of ESRI ArcGIS (ESRI 2012). Watershed areas are given in Table 3-1.

Land use data and uncertainty in the area estimates

To estimate land use area proportions in each watershed, we used land use data from the National Land Cover Database (NLCD) 2006 product (Fry et al. 2011), which identifies fifteen distinct land use classes in our region. We aggregated several of the fine-scale (level II) classes based on two criteria. First, we aggregated land use classes that, from a phosphorus management perspective, should behave similarly (e.g., forest-type classes 41, 42, and 43 and shrub class 52). Second, we grouped some land use classes that had high classification error rates among the classifications, suggesting that aggregation might raise confidence that the map reflected what is actually on the ground (e.g., urban classes 22, 23, and 24, and wetland classes 90 and 95).

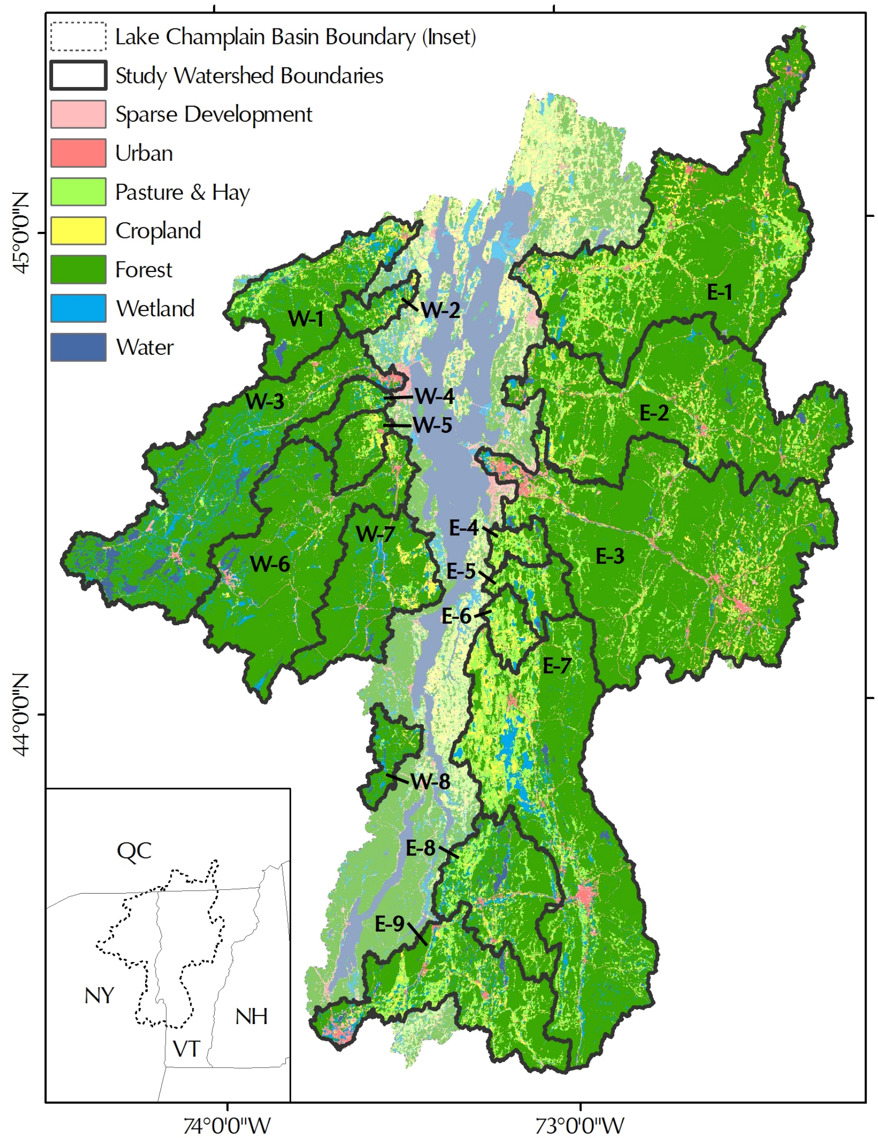

Figure 3-1. Watershed boundaries (bold lines) and land use for the 17 watersheds included in this study. See Table 3-1 for watershed names.

Table 3-1. Watershed areas for the 17 watersheds included in the study. Watershed codes refer to watershed labels in Figure 3-1.

Western Watersheds		Area (km^2)	Eastern Watersheds		Area (km^2)
W-1	Great Chazy	656	E-1	Missisquoi	2203
W-2	Little Chazy	131	E-2	Lamoille	1880
W-3	Saranac	1581	E-3	Winooski	2836
W-4	Salmon	166	E-4	LaPlatte	114
W-5	Little Ausable	183	E-5	Lewis	202
W-6	Ausable	1355	E-6	Little Otter	143
W-7	Bouquet	721	E-7	Otter	2240
W-8	Putnam	155	E-8	Poultney	674
			E-9	Mettawee	958

Using the detailed accuracy assessment done for the NLCD 2006 product (Wickham et al. 2013), and following methods described by Olofsson et al. (2013) and Stehman (2013), we developed bias-adjusted estimates of the area of each land use category for each watershed. These bias-adjusted estimates subtract from each land use area estimate the area of commission and add to it the area of omission, which are quantified in the accuracy assessments. In this way, we adjusted map estimates of the proportional area according to the rate of over- or under-representation in the map relative to the reference imagery, and then estimated confidence limits for those proportions. This information allowed us to express the proportional area of each land use class in each watershed as a normal distribution with a mean equal to the bias-adjusted proportional area estimate, and standard deviation equal to the standard error of that estimate. A normal distribution was appropriate to approximate these because none of the individual proportion estimates were close enough to the upper or lower bounds to risk compressing the distribution.

We also assumed that land use change was insignificant over the period 2006-2013 (the period of analysis for the nutrient generation rates). In some watersheds, change of > 1% per year did occur for some land uses classes, but in most cases, the degree of change was within the bounds of the change vs. no-change classification uncertainty (Jin et al. 2013), and therefore may not reflect actual, on-the-ground changes in land use.

Phosphorus load estimation

The United States Geological Survey (USGS) and the LCBP maintain continuous stream discharge gauges at each water quality sampling location to enable estimates of nutrient delivery to Lake Champlain. We obtained daily average flows for the period 1991 to 2014 from the USGS web-based data warehouse at http://waterservices.usgs.gov.

To support on-going management efforts in the Lake Champlain Basin, stream nutrient concentrations are measured regularly near the outlets of eighteen watersheds through a shared effort of the Vermont and New York State Departments of Environmental Conservation (VTDEC and NYSDEC), the Québec Ministère du Développement durable, de l'Environnment, et des Parcs (QC MDDEP), and the Lake Champlain Basin Program (LCBP). Grab samples are taken 24 times per year in each watershed, with sampling events weighted toward higher flow events. The samples are analyzed for total phosphorus, among other constituents.

We acquired total phosphorus data for each of the study watersheds for the years 1991 through 2013 inclusive in April 2014. We estimated phosphorus loads for each year from 1991 to 2013 using the Weighted Regression on Time, Discharge, and Season (WRTDS) method developed by Hirsch et al. (2010) and applied to the Lake Champlain basin tributaries by Medalie et al. (2012). Under the WRTDS method, nutrient concentrations on any particular day are estimated based on a concentration-discharge relationship where samples taken under similar conditions (i.e., at similar discharges, closer in years, and at a similar point in the year) contribute more to the estimation point than samples taken in dissimilar conditions. See Hirsch and De Cicco (2014) for additional details.

We evaluated the flow-normalized phosphorus flux estimates for the 2006-2013 period, and found that any trend over the period of interest was far overshadowed by year-to-year variability in the loads. Thus we did not build trend detection into our model assuming that year-to-year hydrologic variability, not changes in loading rates, determined actual year-to-year loads for this time period.

Model Structure

We employed a Bayesian hierarchical regression model with varying intercepts (Gelman and Hill 2009) to fit area-normalized phosphorus loads over the period 2006-2013 from proportional land areas in each watershed. We excluded 2011 from the analysis because of a large tropical storm that delivered 13-18 cm of rain over period of about 30 hours. In some watersheds, phosphorus loads in that two-day event surpassed loads for the rest of the year, but that effect was not consistent across the study area. Because we assume that land use is the key driver of phosphorus loads in most years, we excluded that year because of the disproportionate effect of that single storm event.

Hierarchical models allow a hybrid approach between two alternative model forms – a complete pooling method where all watersheds are observations in the same model and variance is pooled across all watersheds, and a no-pooling method where watersheds are broken into groups based on similarity and models are developed separately for each group (Qian et al. 2010). This approach strikes a compromise between these two approaches by allowing information flow between groups while retaining the ability to predict group-level differences.

Watershed groups were delineated by clustering using the relationship between the average area-normalized load and the standard deviation in area-normalized load for each watershed. We assumed that the area-normalized loads for each watershed were normally distributed, with mean equal to the sum of a watershed-specific mean and a group-level factor, with group-level variance:

$$L_i \sim N(\alpha_g + \mu_i, \sigma_g)$$

where L_i is the area-normalized load in watershed *i*, α_g is the intercept for group *g*, μ_i is the mean area-normalized load in watershed *i*, and σ_g is the variance of group *g*. Area-normalized load for each watershed (i.e., the data-level model) was predicted by the sum of the products of the proportion of each land use and the nutrient generation rate:

$$\mu_i = \sum_{j=1}^{J} EC_j * P_{i,j}$$

where EC_j is the export coefficient for land use *j*, and $P_{i,j}$ is the proportion of land use *j* in watershed *i*. A factor representing group memberships was the only predictor for the group-level parameters α and σ. While other implementations of ECMs have included a term to account for watershed-scale nutrient loss (through sedimentation, for example), two previous studies have determined that watershed attenuation of phosphorus is minimal in our region, and therefore we opted to leave out a coefficient to represent that specific process (Budd and Meals 1994, Preston et al. 2011).

To incorporate the error in the land use classification, we expressed the estimate for $P_{i,j}$ as a normal distribution with mean at the bias-adjusted map estimate and standard deviation equal to the standard error of the estimate. For each iteration of the model, we drew a sample from the distributions for each land use proportion in each watershed, and fit coefficients to that estimate.

Prior distributions for the land use coefficients were developed from ongoing modeling that is part of a Total Maximum Daily Load revision for Lake Champlain. Here, land use loading rates were generated by the Soil and Water Assessment Tool (SWAT) for each basin (Tetra Tech 2013b). We calculated mean loading rates across basins and soil types from that model, to use as the mean loading rates for our priors. Tetra Tech also reported interquartile ranges for their loading rates, and we calculated standard deviations from those as an approximation of the spread in loading rates. To provide more flexibility in our model, we defined the standard deviation of our priors at five times the standard deviation calculated from the Tetra Tech loading rates. Additionally, all land use coefficients other than those for water and wetland were truncated at zero, to accommodate the theoretical constraint that those land uses could not act as phosphorus sinks. For the group level parameters α and σ we used uninformative normal and lognormal priors, respectively.

We approximated the posterior distributions of the group intercepts, variance parameters, and land use coefficients via Markov Chain Monte Carlo implemented in the statistical software JAGS (Plummer 2013) and R (R Core Team 2014). We ran the model through 250,000 iterations, and retained every 50th posterior sample for the two group-level parameters and the seven land use classes. Post-processing of model output was done in the R package rjags (Plummer 2014), and results were plotted using the R packages ggplot2 (Wickham 2009).

In most applications of export coefficient models of this kind, the regression line is forced through the origin to accommodate the theoretical consideration that a watershed of size zero should have a nutrient load of zero. However, because this model predicted area-normalized loading from proportional land uses, we interpreted the intercept as watershed-scale effects on the load that could not be explained by land use patterns. Therefore, we did not force the intercept through zero. Examples of many effects that behave in this way have been documented in the literature. The presence of reservoirs, watershed topography, and historical land use are just three of a long list of factors that could produce observable differences in delivered watershed loads.

Results & Discussion

Land Use Areas

Bias adjusted land use proportions are shown in Figure 3-2. Forest cover dominates most of the area (74.6%), but anthropogenic uses (i.e., agricultural and developed land uses) together account for 20.6% of the land use across watersheds. The bias adjustment made important changes to the areal estimates of individual land use classes; relative to the reference imagery, sparse development was under-represented in the original map by more than 48%, while wetlands were over-represented by more than 61% (Figure 3-2). Because our model relied on land use proportions rather than the raw area, these changes are large in relation to the scale of the data.

Watershed P Loads

Watershed total phosphorus loads are shown in Table 3-2 for the years 2006-2013, exclusive of 2011. Bias in the WRTDS model estimates of daily flux was low for all watersheds – only one watershed had an average bias of more than 5% in absolute value. Because the uncertainty in each year's loading estimate was greatly overshadowed by the annual variability associated with hydrologic differences, we ignored uncertainty associated with the annual load estimates.

Table 3-2. Total phosphorus loads for the 17 watersheds, 2006-2013 (exclusive of 2011), in metric tons (1000 kg). Percent bias of the WRTDS model is calculated as the bias in estimated daily flux relative to observed daily flux.

	Total Phosphorus Load (metric tons)								**% Bias (WRTDS)**
	2006	2007	2008	2009	2010	2012	2013	*Mean*	
Ausable	43.2	37.6	37.3	22.0	28.5	21.8	37.2	32.5	0.2
Bouquet	32.5	22.9	28.2	14.2	14.4	13.0	26.7	21.7	3.3
Great Chazy	29.5	20.4	21.2	20.0	15.5	14.5	20.2	20.2	0.3
Lamoille	97.7	56.8	63.2	36.4	39.8	26.7	65.6	55.2	-1.9
LaPlatte	11.0	7.0	8.6	3.3	4.2	2.8	9.7	6.7	2.1
Lewis	19.7	11.4	11.7	6.3	9.7	4.3	11.1	10.6	-2.8
Little Ausable	4.1	3.1	4.1	2.1	2.5	2.5	5.7	3.5	-7.7
Little Chazy	6.6	5.0	5.5	3.2	2.9	2.7	6.3	4.6	-1.5
Little Otter	13.7	8.9	16.7	8.4	7.7	5.6	11.7	10.4	0.5
Mettawee	39.3	25.6	36.1	24.1	21.7	14.7	20.2	26.0	4.4
Missisquoi	288.4	198.8	174.2	92.5	138.1	99.6	160.9	164.6	-0.7
Otter Creek	140.9	99.5	122.8	117.0	85.4	58.5	85.6	101.4	-0.4
Poultney	41.3	34.6	43.3	27.9	24.8	11.9	24.8	29.8	-1.1
Putnam	2.8	2.1	1.9	1.8	1.9	1.6	2.3	2.0	-12
Salmon	2.8	2.3	3.3	1.6	1.8	2.1	3.8	2.5	0.1
Saranac	31.9	27.5	31.9	24.8	19.0	19.3	33.9	26.9	-4.8
Winooski	297.5	205.8	234.6	102.4	118.2	73.2	237.4	181.3	1.6

Area-normalized loads were used as the dependent variable in the model in part to satisfy the normality assumptions of linear regression. Loads for all years were divided by the watershed area. The relationship between average area-normalized load and the standard deviation determined group membership (Figure 3-3). The ordering appeared largely, but not entirely, independent of the size of the watershed.

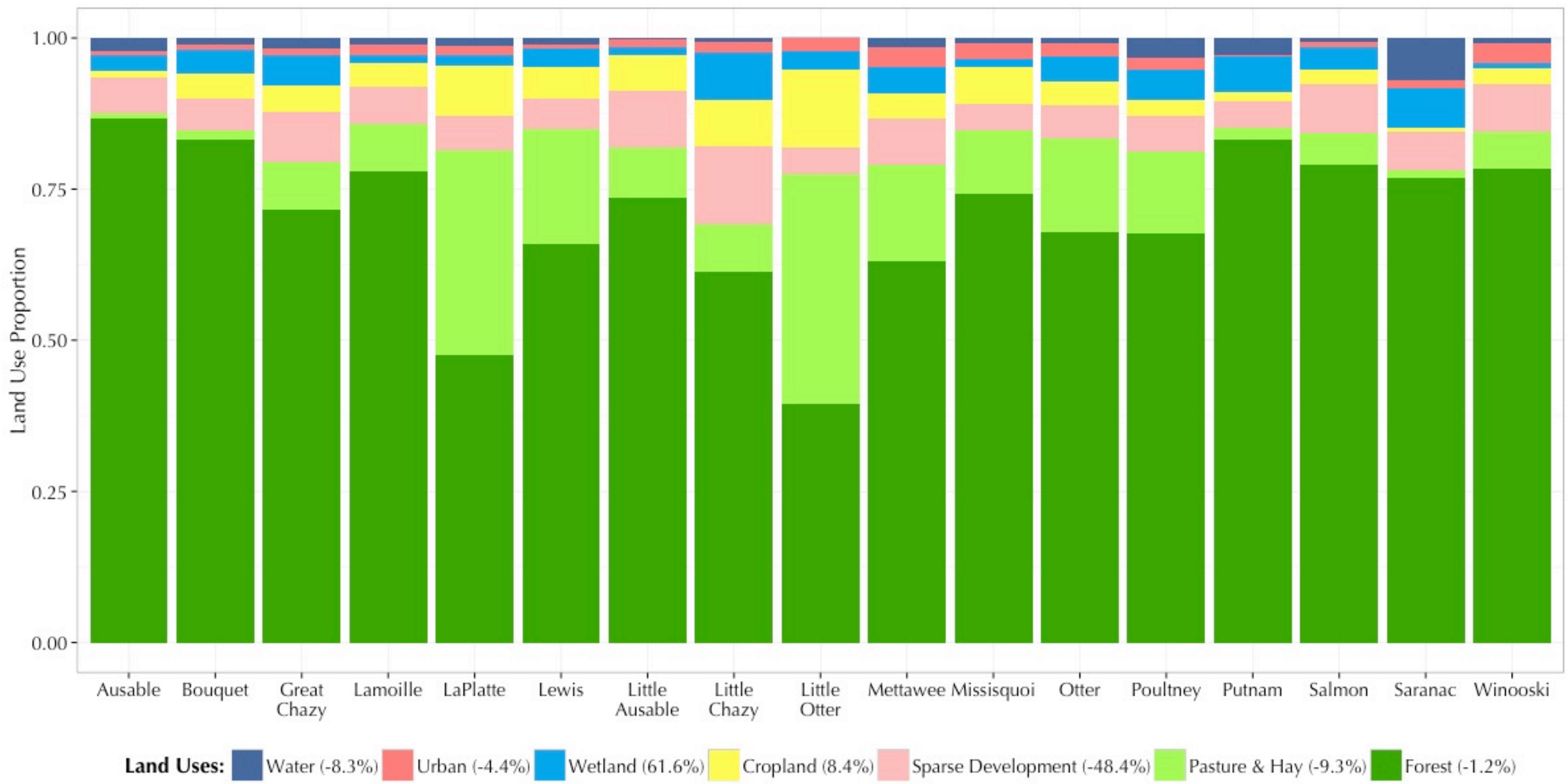

Figure 3-2. Bias adjusted land use proportions for each watershed included in the study. Bias adjustment rates for each land use are indicated in parentheses in the legend. Negative and positive values indicate under- and over-representation, respectively, relative to the original map.

Model Diagnostics

Convergence diagnostics for the model were favorable. Traceplots showed that mixing between the five chains was good over the iterations (Figure 3-4,Figure 3-5), and Rhat (an estimate of convergence) approached 1.000 for all variables in the model. Autocorrelation and cross-correlation plots also showed no signs of poor convergence or other problems.

Land use phosphorus generation rates

Medians and confidence intervals for posterior land use coefficient samples are given in Table 3-3. Prior and posterior density functions for the land use phosphorus generation rates (Figure 3-6), show that in most cases, posterior means were less than prior means (with the exception of the Forest land use), and the variance was smaller in all cases. This outcome was expected because SWAT generated loading rates are intended to represent the "edge-of-field" loading rates, before any stream processing effects. In contrast, export coefficient models such as ours use delivered watershed loads to estimate the proportional contribution of land uses at the end of the tributary, after the effect of any in-stream processes.

Table 3-3. Median and 95% confidence intervals for posterior samples from the export coefficient model. All estimates are in kg/ha/yr.

Interval	Water	Urban	Sparse Development	Forest	Pasture & Hay	Cropland	Wetland
2.5%	-0.83	0.31	0.12	0.03	0.01	0.07	-0.23
Median	0.67	2.17	0.82	0.23	0.21	0.91	0.12
97.5%	2.75	4.46	1.93	0.50	0.76	2.55	0.48
Std. Dev.	0.89	1.09	0.47	0.13	0.20	0.66	0.18

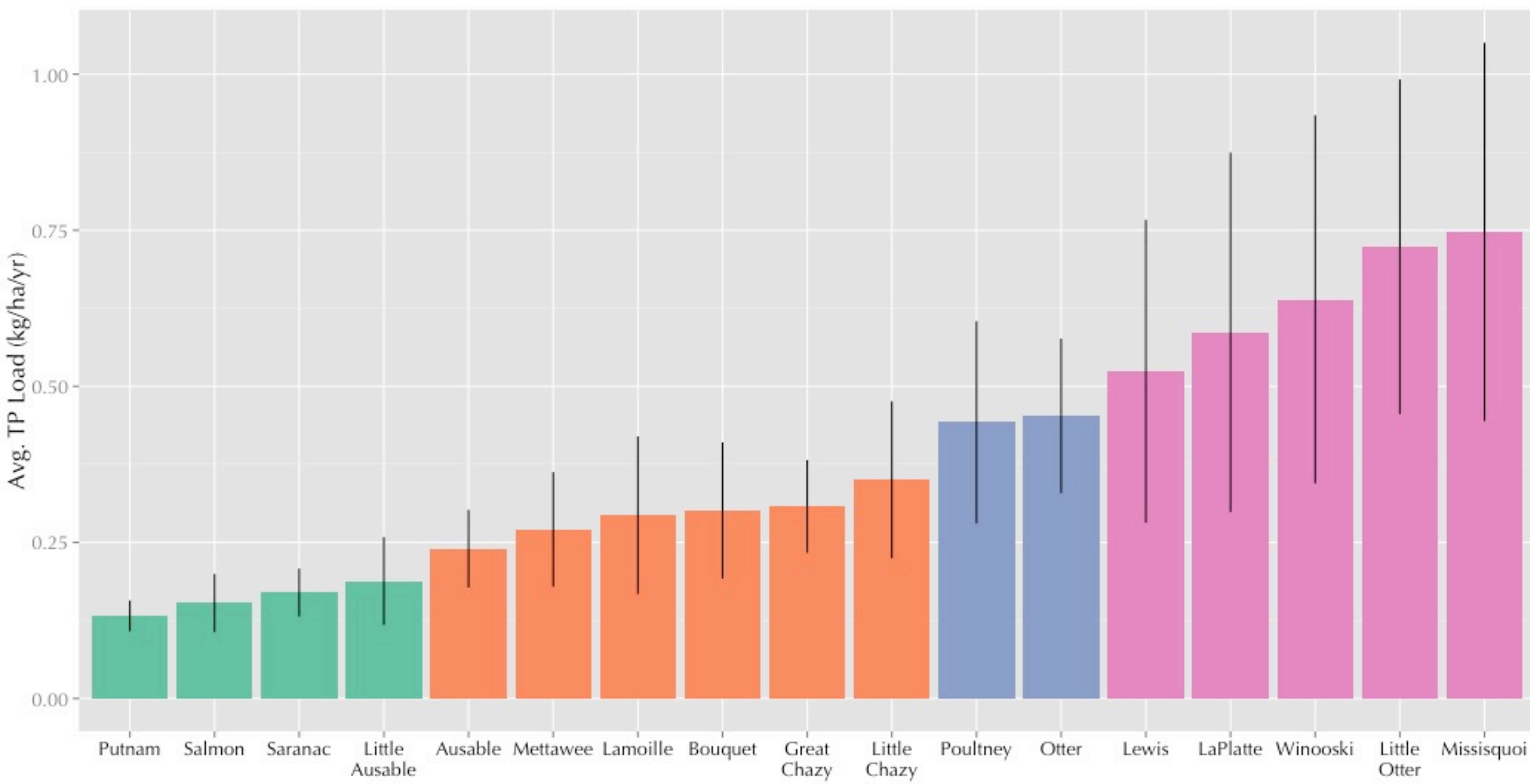

Figure 3-3. Average area-normalized loads (bars) and standard deviation (bars), for each watershed as estimated by the WRTDS model. Colors show group membership used in the hierarchical model.

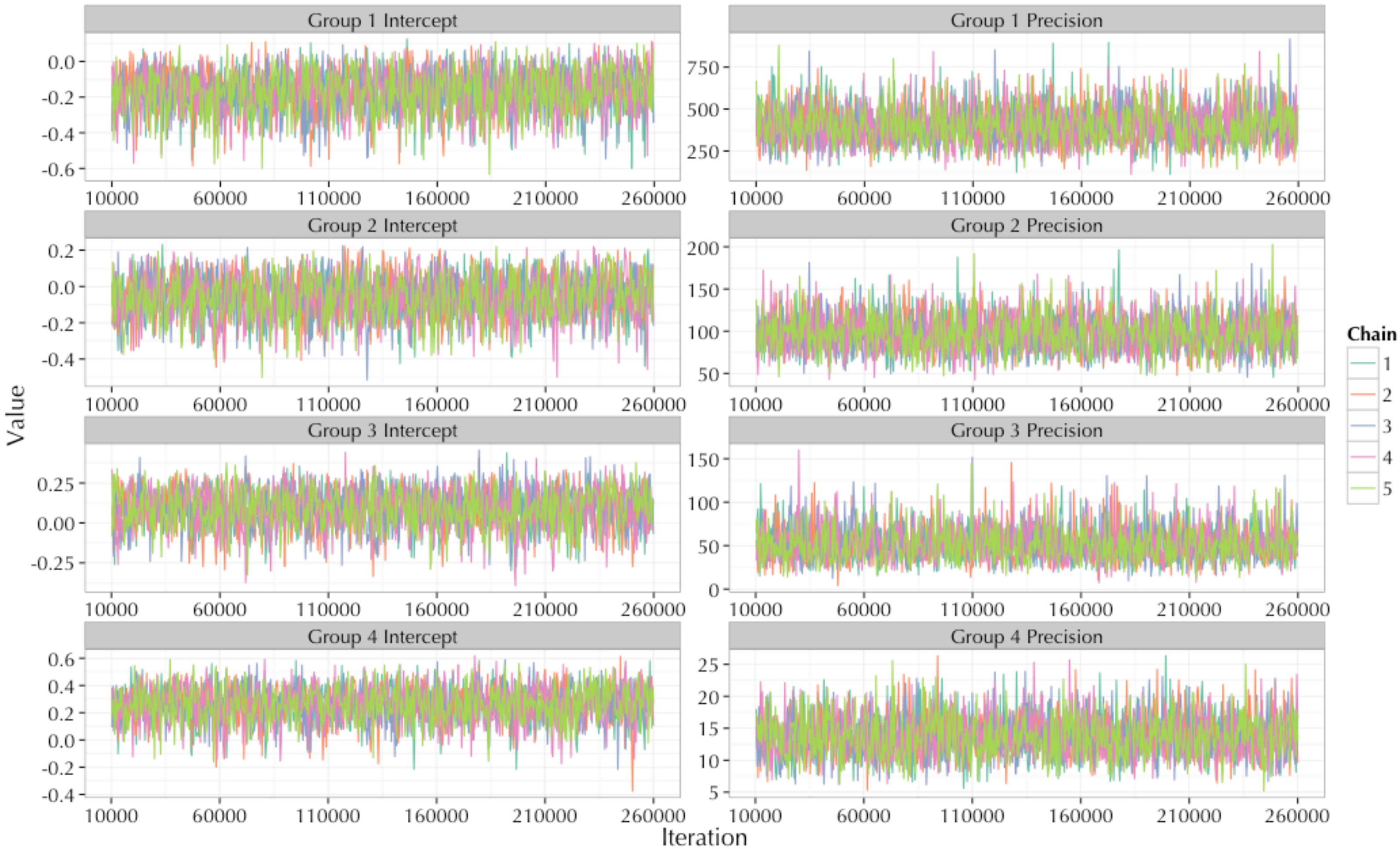

Figure 3-4. Traceplots for the group-level model parameters from the post-burnin period, showing good mixing between model chains. Values for intercepts are in units of kg/ha/yr, while values for precision parameters are 1/variance.

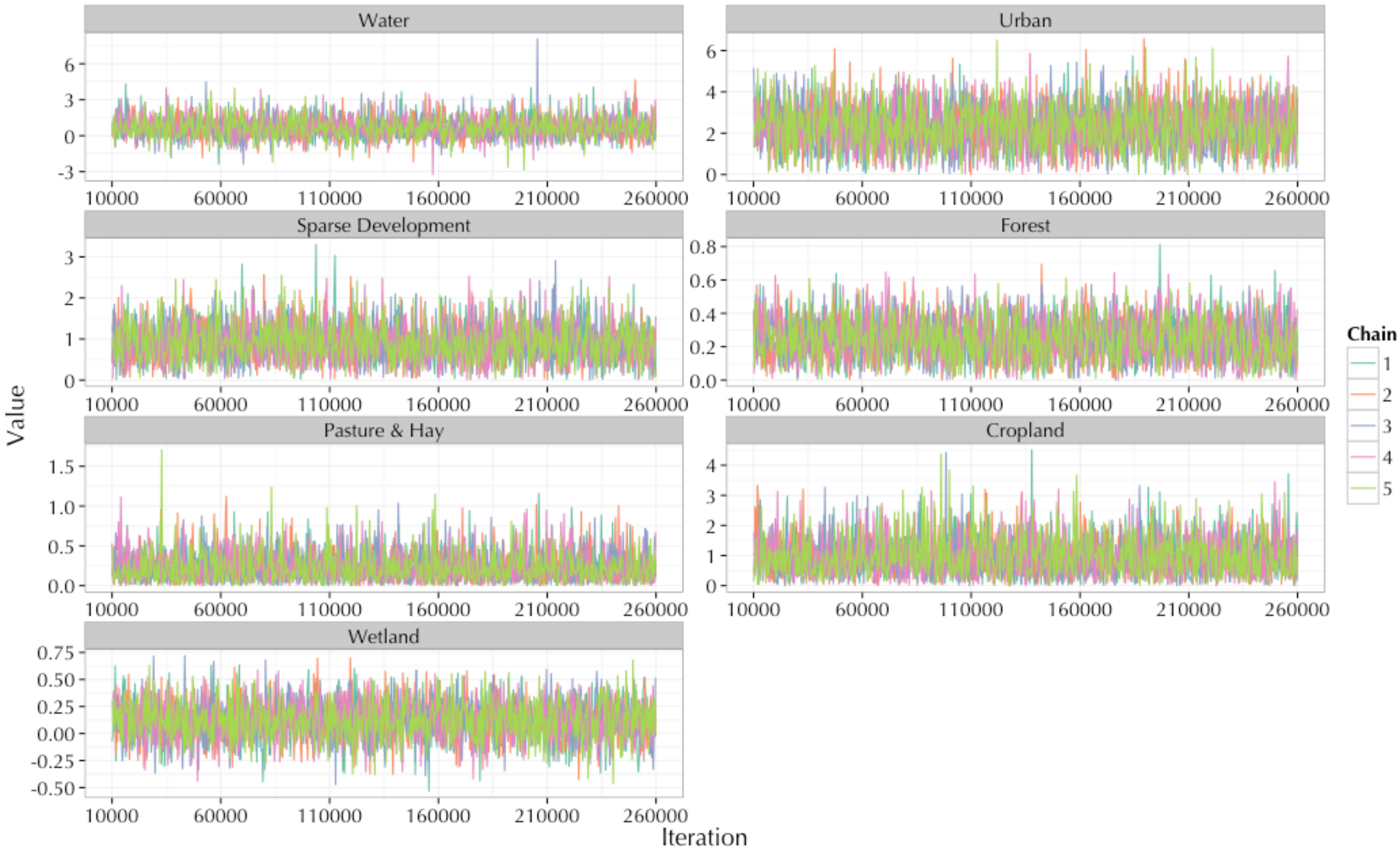

Figure 3-5. Traceplots for land use coefficients from the post-burnin period, showing good mixing between model chains. Values for all coefficients are in units of kg/ha/yr

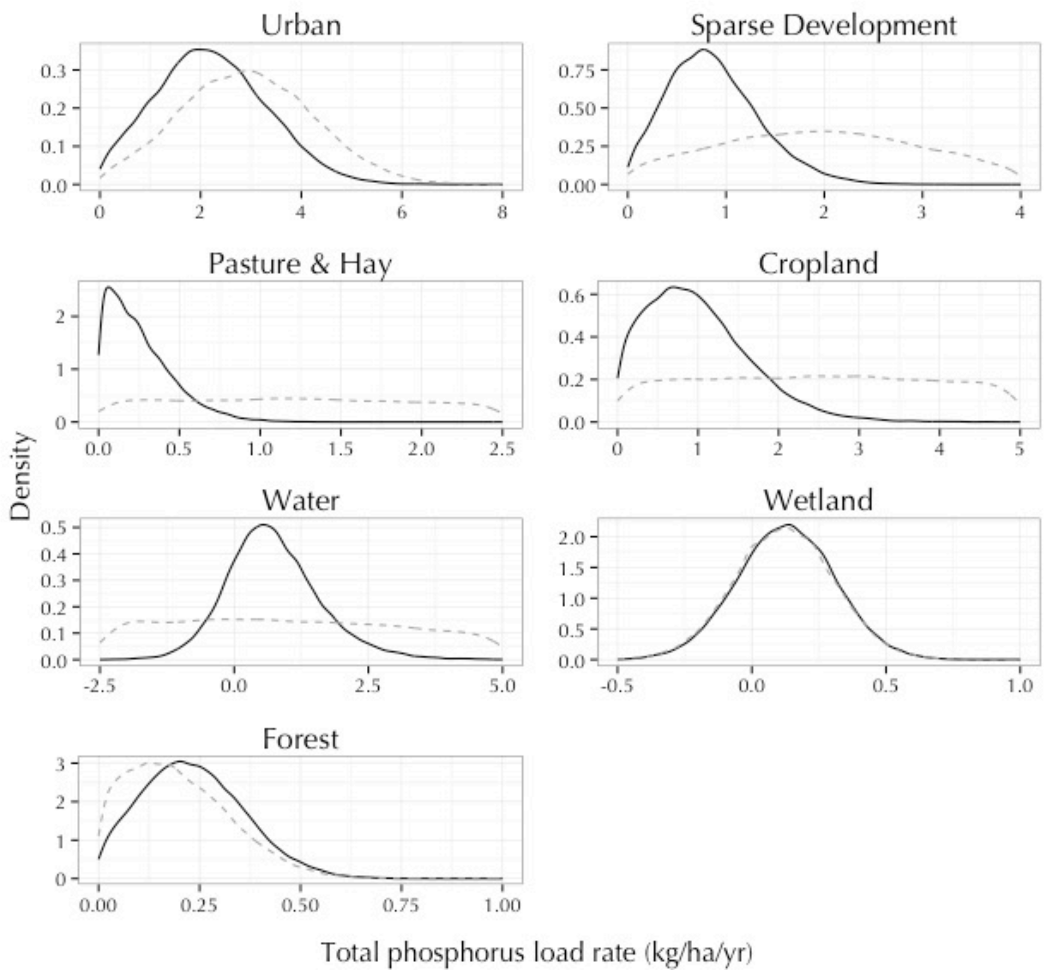

Figure 3-6. Prior (dotted line) and posterior (solid line) density functions for the estimates of the land use specific phosphorus generation rates.

Posterior estimates for the intercepts also departed from their (single) prior distribution (Figure 3-7). Intercept effects are largely negative for groups 1 and 2 (i.e., most of the density function lies below 0), but largely positive for groups 3 and 4. In the calculation of total loads for the watersheds, this indicates that the watersheds in groups 1 and 2 are exporting less phosphorus than explained by land use (i.e., attenuating phosphorus). While these data provide no direct insight into the drivers behind likely attenuation at the watershed scale, combinations of effects such as extensive wetland networks, acidic soil chemistry, and the presence of reservoirs may play a part in determining group membership for at least some of the watersheds in those groups. On the other end of the spectrum, intercept values for groups 3 and 4 indicate that those watersheds export more phosphorus than explained by land use alone. We interpret these additional loading effects as evidence of residual loading, potentially caused by historically high fertilizer use (Ford 2012), or by erosion from unstable stream channels. Evidence the latter has been found in the Lake Champlain Basin using data collected along 1400 miles of stream channel in Vermont (Kline and Cahoon 2008, Langendoen et al. 2012, Tetra Tech 2013b).

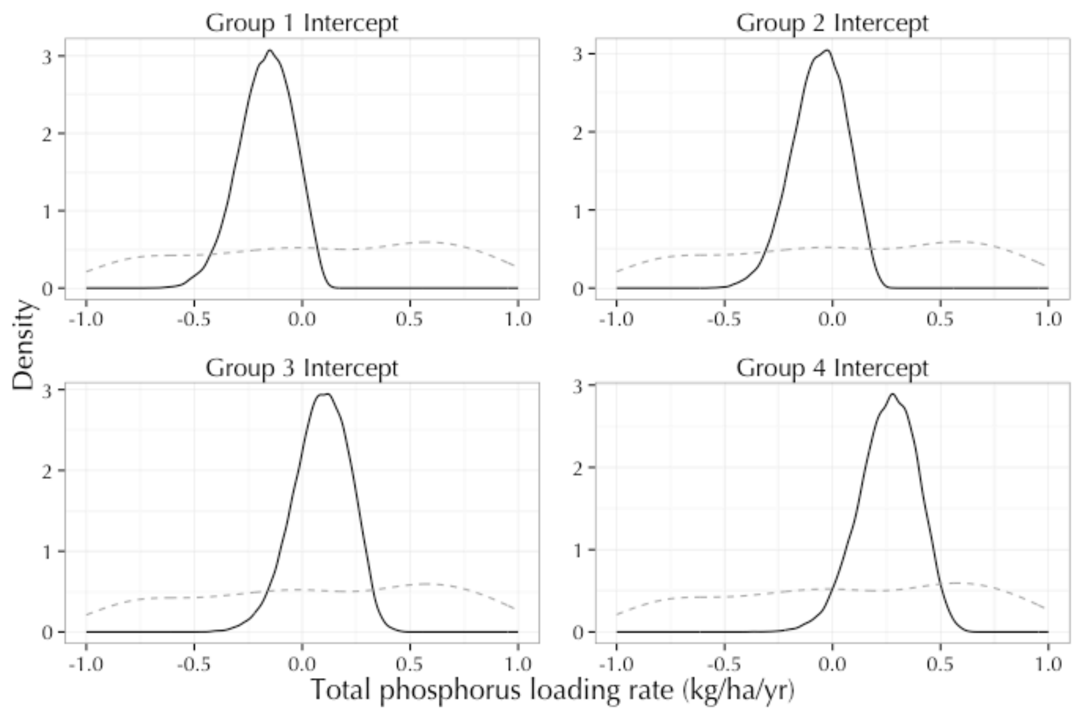

Figure 3-7. Prior density (dotted line) and posterior density (solid line) functions for intercept values from the export coefficient model.

One important advantage of the Bayesian statistical framework is the ability to propagate variability and uncertainty among model parameters. Of particular interest to us was the ability to propagate variability from the loading data, which is driven primarily by year-to-year hydrologic variability, to the estimates of the intercepts and loading rate coefficients. The uncertainty associated with the loading rate coefficients is derived from multiple sources including uncertainty in the land classification, variability in the watershed loads, and random error, and their interactions and relative contributions are not easy to parse out. However, the variability associated with the intercept comes only from variability in the load and the group-level model error. Therefore, the variability in the intercepts can more easily (albeit still with caution) indicate the effect of hydrologic variability in a year-to-year phosphorus load generation.

Phosphorus Load Prediction and Model Validation

To predict total phosphorus load for each watershed, we sampled from the joint posterior distributions of each land use coefficient, appropriate group-level intercept and model precision parameter over 1000 iterations using the covariance matrix obtained from the model. We then calculated the area-normalized load using the model form described above, and multiplied that by the watershed area. Over 1000 iterations, that allowed us to calculate a mean and variance associated with those predictions and to compare those to the observed means and variances for each watershed. Figure 3-8 and Figure 3-9 show the predicted versus observed area-normalized loads and total loads, respectively, with all watersheds included.

Because of the relatively small sample size of watersheds and years of loading data, we opted for a leave-one-out approach to model validation. We ran the model excluding one watershed at a time, sampled from the posterior coefficient estimates to predict the loads for all watersheds, and then calculated the Nash Sutcliffe Efficiency (NSE) for that set of predictions. We checked the ability of these coefficients to predict the mean area-normalized watershed load, the total watershed load (area-normalized load multiplied by the watershed area), and the variance in each estimate for each watershed against observed data for the period of analysis. The average NSE was 0.889 for the area-normalized model, and 0.968 for the scaled model.

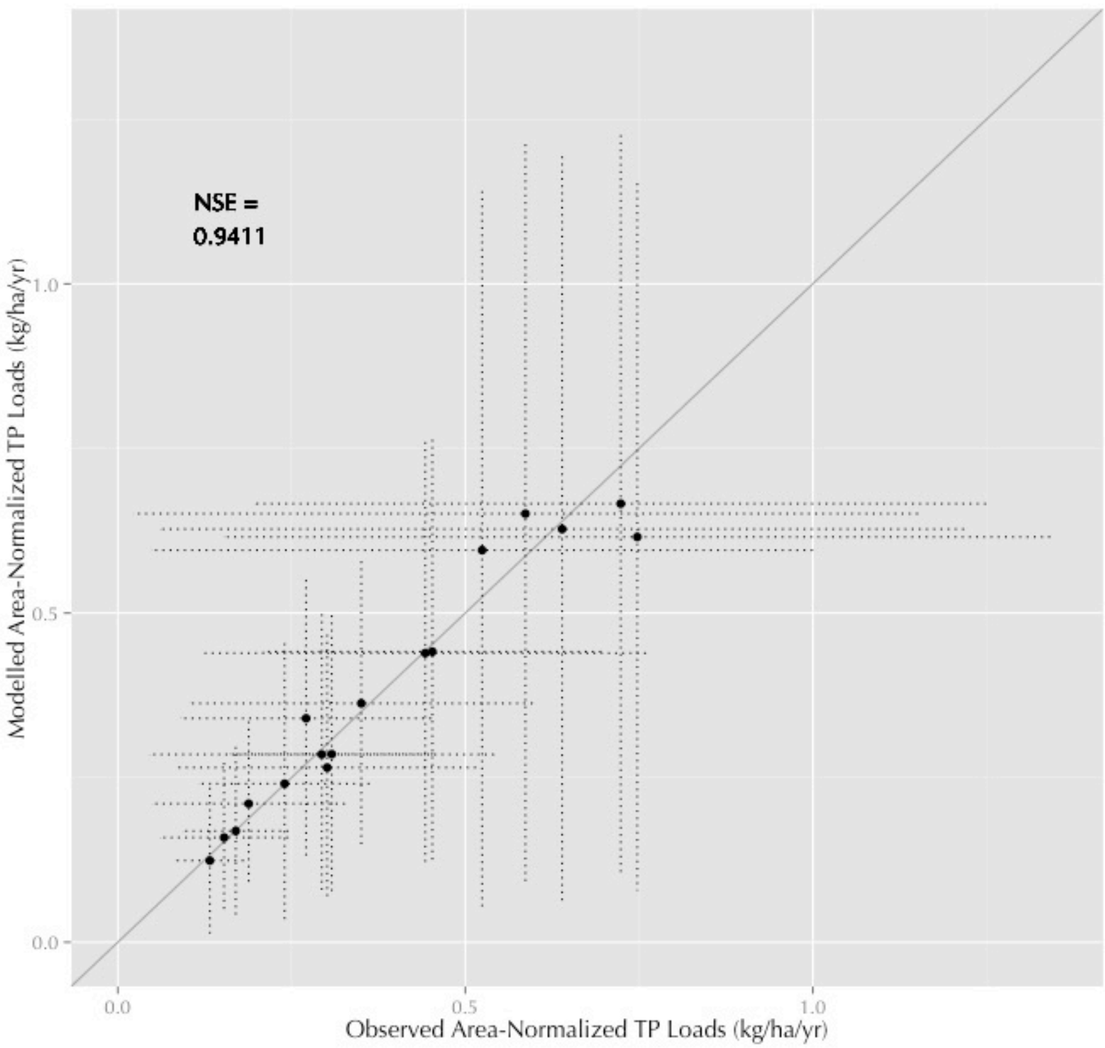

Figure 3-8. Predicted versus observed average area-normalized loads (dots) and 95% prediction intervals (dotted lines) for the 17 watersheds over the period 2006-2013. Grey diagonal line is the 1:1 line. NSE=Nash Sutcliffe Efficiency

Land Use-Specific Loads by Watershed

Land use loads for each watershed (Table 3-4) include a much higher contribution from forested lands in the Lake Champlain Basin than has been estimated in the past (e.g., Budd and Meals 1994, Hegman et al. 1999, Troy et al. 2007), but they are generally in line with more recent loading estimates (Tetra Tech 2013). Our results also indicate that in some watersheds, residual loading can add a significant proportional load, and that in others, the watershed has strong assimilative capacity (shown by negative residual loading). Because loading from forest lands and residual loading are generally considered unmanageable sources of phosphorus, these land use specific loads have important consequences for the ability of natural resource managers to attain watershed-specific targets.

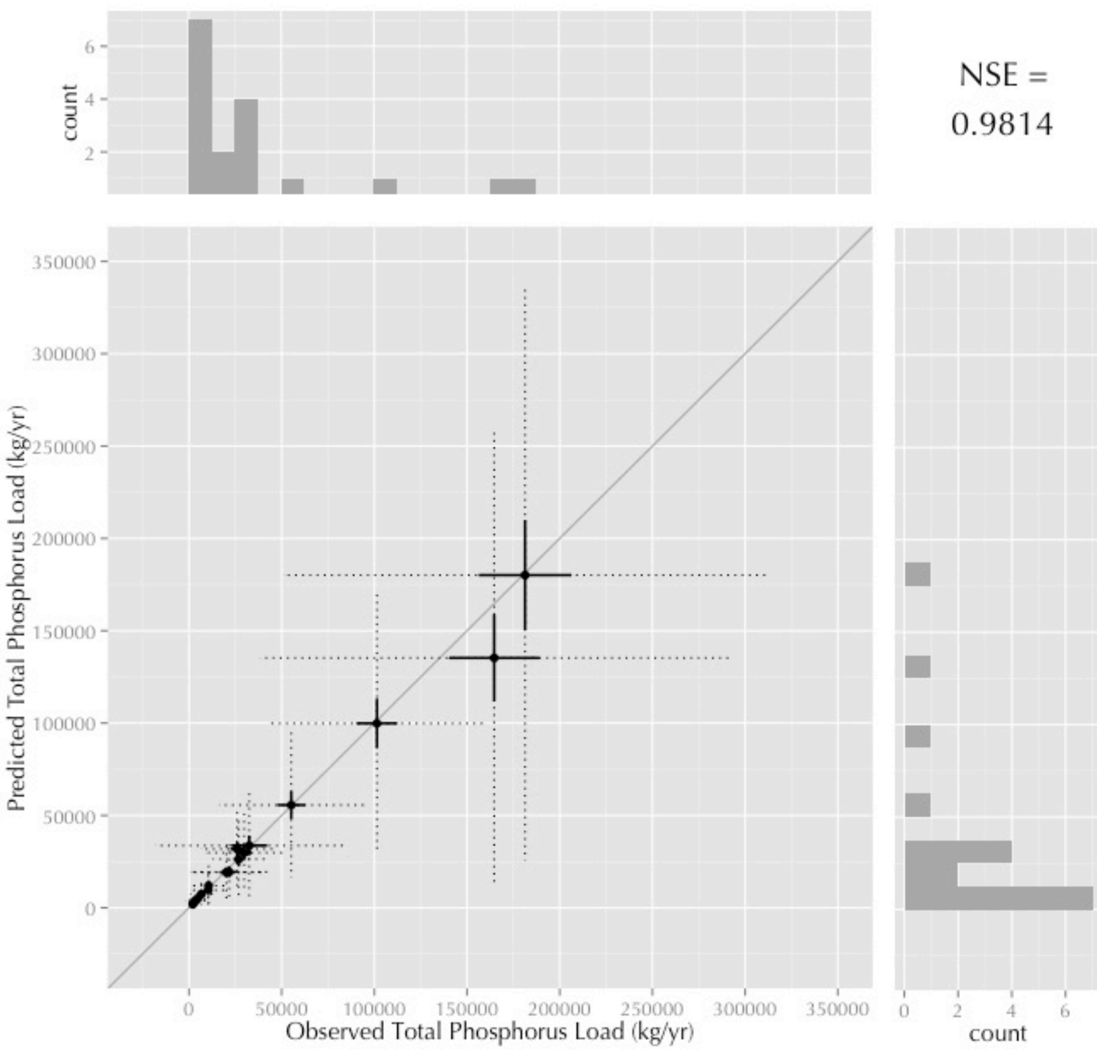

Figure 3-9. Predicted versus observed average total loads (dots), standard errors (dark lines), and 95% prediction intervals (dotted lines) for all watersheds over the period 2006-2013. Grey diagonal is the 1:1 line. NSE=Nash Sutcliffe Efficiency.

Table 3-4. Total phosphorus load predicted by the export rates estimated from the hierarchical model, using bias-adjusted land use areas and their uncertainty in the predictions. All values are in mt/yr. Negative values indicate phosphorus attenuation. % Bias is calculated from observed TP loads 2006-2013.

	Total Phosphorus Load, metric tons per year									
Watershed	Water	Urban	Sparse Development	Forest	Pasture & Hay	Crops	Wetlands	Residual	Total	% Bias
Ausable	2.4	2.5	7.3	29.7	0.4	1.4	0.5	-11.0	33.2	-0.27
Bouquet	0.6	1.5	3.5	15.2	0.3	3.2	0.4	-5.9	18.8	-11.80
Great Chazy	0.9	1.6	5.0	11.9	1.6	3.0	0.4	-5.3	19.1	-5.80
Lamoille	1.7	7.5	10.5	37.2	4.4	8.0	0.3	-15.3	54.3	-0.33
LaPlatte	0.1	0.4	0.6	1.4	1.2	1.0	0.0	2.7	7.3	11.27
Lewis	0.2	0.3	0.9	3.4	1.2	1.1	0.1	4.7	11.9	9.18
Little Ausable	0.0	0.5	1.6	3.4	0.5	1.2	0.0	-3.4	3.8	11.47
Little Chazy	0.1	0.5	1.6	2.0	0.3	1.0	0.1	-1.1	4.7	0.58
Little Otter	0.0	0.7	0.6	1.4	1.6	2.0	0.1	3.3	9.7	-7.26
Mettawee	1.2	7.4	6.7	15.3	4.6	4.3	0.5	-7.8	32.2	22.59
Missisquoi	1.7	12.4	9.2	41.5	6.8	14.1	0.4	51.4	137.5	-16.37
Otter	1.6	11.3	11.4	38.5	10.5	9.3	1.2	16.1	99.9	-0.23
Poultney	1.8	3.1	3.6	11.6	2.7	1.8	0.4	4.8	29.9	0.68
Putnam	0.4	0.1	0.6	3.3	0.1	0.3	0.1	-2.9	1.9	-5.42
Salmon	0.1	0.3	1.2	3.3	0.3	0.4	0.1	-3.1	2.6	1.16
Saranac	9.5	4.1	9.6	30.8	0.6	0.7	1.4	-29.4	27.3	0.50
Winooski	2.1	20.9	20.7	56.3	5.2	8.2	0.3	66.2	179.8	0.18

Comparison with traditional ECM methods

We compared results from our model with results from previous export coefficient studies performed in the Lake Champlain basin using standard frequentist approaches (Budd and Meals 1994, Hegman et al. 1999, Troy et al. 2007). In each of the previous studies, multiple regressions using untransformed areas of aggregated urban, agricultural and forested land uses were used as predictors, and untransformed phosphorus loads as responses. Each study also forced the regression through the intercept. Interestingly, each study also made corrections to the loading coefficients for heavily agricultural watersheds because of their status as outliers, which was not required in this study.

Because of the different time periods associated with each study, direct comparisons of total load were not informative. However, we compared the loading coefficients and proportional contribution of loading sources from each study (Table 3-5). To enable more direct comparisons between our study and these previous works, we aggregated results from urban and agricultural land uses; other land uses (i.e., forest, water, and wetland) and residual loads, which were aggregated into an "unmanageable" category to reflect the assumption that these categories are generally not subject to management.

The coefficients associated with urban and agricultural land uses are roughly similar across studies, and though the coefficient associated with forested lands is much greater in our study, our value was well within ranges reported in the literature reviews performed by Budd and Meals (1994) and Hegman et al. (1999). The largest difference between our study and these previous studies is the relative contribution of the various sources, particularly the unmanageable category. The primary reason for this is the inclusion of a term (i.e., the regression intercept) to account for watershed-scale residual loading, which makes up a large proportion of the total load in some watersheds.

Table 3-5. Comparison of phosphorus loading coefficients and proportion land use contributions estimated in this study with those of previous studies

	Budd and Meals (1994)	Hegman et al. (1999)	Troy et al. (2007)	This study
Land Use Loading Coefficients (mt/yr)				
Urban	1.5	1.5	2.5	1.55
Agriculture	0.5	0.42	0.61	0.53
Forest	0.1	0.04	0.04	0.24
Proportional Load (%)				
Urban	16.8	37.1	53.1	26
Agriculture	56.8	51.3	39.3	14.7
Unmanageable	26.5	11.6	7.7	59.3

Table 3-6. Comparison of land use loading rate estimates from model forms with and without bias-adjusted land use area estimates and propagation of uncertainty in land use classifications

Statistic	Water	Urban	Sparse Development	Forest	Pasture & Hay	Cropland	Wetland
Bias-Adjusted Land Uses and Uncertainty							
Mean	0.64	2.29	0.91	0.25	0.24	0.97	0.12
Median	0.67	2.17	0.82	0.23	0.21	0.91	0.12
Std. Dev.	0.88	1.09	0.47	0.13	0.19	0.63	0.18
Bias-Adjusted Land Uses Only							
Mean	0.89	1.91	1.05	0.31	0.19	1.17	0.13
Median	0.89	1.88	1.05	0.31	0.15	1.13	0.14
Std. Dev.	0.52	0.98	0.42	0.13	0.16	0.63	0.18
Unadjusted Land Uses Only							
Mean	0.93	1.95	1.54	0.28	0.17	1.07	0.16
Median	0.92	1.91	1.53	0.28	0.13	1.03	0.15
Std. Dev.	0.59	0.98	0.58	0.12	0.14	0.56	0.17

We also compared the estimates of the land use loading rates and total land use loads generated by our model to estimates generated from using unadjusted land use areas and from ignoring land use classification errors (Table 3-6, Table 3-7, Table 3-8). Though we did not see any consistent evidence that ignoring classification error led to biasing of regression coefficients toward zero, in general, loading rate coefficients estimated under uncertainty did have higher variance, as expected. However, the scale of the difference is commensurate with the degree of uncertainty in the classification – little uncertainty exists for areal estimates of aggregated (level I) land use classes such as forest, whereas more uncertainty exists for level II classes such as Sparse Development and Pasture & Hay (Wickham et al. 2013).

Conclusions

This study adds two significant methodological developments to the literature on export coefficient models. First, the use of the hierarchical structure in a Bayesian framework allowed us to use variable annual loading data and uncertain land use data to estimate land-use specific phosphorus generation rates that both produce reliable estimates of total loads across watersheds and replicate the observed within-watershed variability in annual loading rates. The Bayesian framework allowed us to include and efficiently propagate important sources of uncertainty and variability to other parameters of interest in the model.

Second, many previous export coefficient applications have removed the intercept from the export coefficient model and forced the regression through the origin to accommodate the theoretical consideration that watersheds of size zero should have no phosphorus export (Budd and Meals 1994, Hegman et al. 1999, Shrestha et al. 2008, Liu and Lu 2013). We opted to include the intercept for statistical reasons (the origin acts as a high-leverage point far from the mean of the data), but more importantly, to allow for the effects of watershed-to-watershed differences on phosphorus loading to surface in the model. This analysis shows that these effects can introduce large, uncontrollable sources of phosphorus, which in turn have significant implications for the ability of resource managers to achieve watershed targets for phosphorus loads.

Export coefficients and land use-specific loading estimates often provide the foundation for analysis of alternative management strategies and in setting management targets. However, there is often significant uncertainty in the base data used to generate those estimates, and in the assumptions that determine which predictors to include or exclude. We demonstrate here that incorporating and propagating that uncertainty and making fewer assumptions through a Bayesian hierarchical approach yields valuable insights into the sources NPS phosphorus pollution and the ability of managers to control phosphorus loading and restore water quality.

Table 3-7. Differences in land use loading estimates (mass and percent) between the model form we present in this study and the standard Bayesian formulation using bias-adjusted land use estimates but ignoring uncertainty. Negative values indicate lower predictions by the model form we present.

	Water		Urban		Sparse Development		Forest		Pasture & Hay		Crops		Wetlands		Residual	
Watershed	mass	%	mass	%	mass	%	mass	%	mass	%	mass	%	mass	%	mass	%
Ausable	-0.3	-10.6	0.4	16.2	-1.1	-15	-9	-30.3	0.1	17.2	-0.2	-17.1	0	3.1	8.5	-76.9
Bouquet	-0.1	-18.5	0.2	15.9	-0.5	-14.4	-4.6	-30.2	0.1	24.4	-0.5	-15.3	0	5.3	4.4	-75.4
Great Chazy	-0.2	-17.3	0.3	17.9	-0.8	-16.4	-3.6	-30	0.4	27	-0.4	-12.9	-0.1	-15.2	4.1	-77.7
Lamoille	-0.2	-13.1	1.1	15.2	-1.6	-14.8	-11.2	-30.1	1.1	25.1	-1.2	-14.7	0	-4.7	11.7	-76.5
LaPlatte	0	-33.2	0.1	14.4	-0.1	-16.9	-0.4	-27.5	0.3	27.9	-0.1	-13.1	0	0	0.5	19.4
Lewis	0	2.2	0	2.5	-0.2	-18.9	-1	-29.3	0.3	27.9	-0.2	-15.9	0	15.7	0.8	17.7
Little Ausable	0	0	0.1	15.8	-0.2	-15.6	-1	-30.8	0.2	31.8	-0.1	-9.8	0	0	1.3	-38.9
Little Chazy	0	26	0	8.5	-0.2	-14.3	-0.7	-32.8	0.1	24.2	-0.2	-19.7	0	-38.1	0.8	-71.2
Little Otter	0	0	0.1	20.1	-0.1	-15.3	-0.5	-33.7	0.4	23.6	-0.2	-11.3	0	41.6	0.6	16.8
Mettawee	-0.1	-7.8	1.1	15.1	-1	-15.5	-4.6	-30.3	1.2	25.3	-0.6	-13.7	-0.1	-10	6	-76.4
Missisquoi	-0.2	-12.7	1.9	15.4	-1.4	-14.7	-12.5	-30.1	1.7	24.4	-1.9	-13.7	0	-0.3	9.3	18
Otter	-0.1	-8	1.7	15.3	-1.8	-15.5	-11.6	-30.1	2.6	25.1	-1.4	-14.6	0	-2.6	11.5	71.4
Poultney	-0.2	-11.5	0.5	15.8	-0.6	-16.5	-3.4	-29.7	0.6	23.8	-0.3	-16.3	-0.1	-15.7	3.4	71.2
Putnam	0	0.6	0	25	-0.1	-17.5	-0.9	-28.5	0	27.6	0	-0.3	0	-24.4	1.1	-37.4
Salmon	0	-13	0.1	25.8	-0.2	-19.5	-1	-30.7	0.1	34.5	0	-11.8	0	12.5	1.2	-37.5
Saranac	-1.1	-11.3	0.7	16	-1.5	-15.2	-9.2	-29.9	0.1	23.9	-0.1	-21.4	0	-2.9	11.3	-38.5
Winooski	-0.2	-10.3	3.2	15.5	-3.1	-15.2	-17	-30.2	1.3	25.4	-1.1	-13.9	0	-1.4	11.9	18
Mean	*-0.2*	*-8.1*	*0.7*	*15.9*	*-0.9*	*-16.0*	*-5.4*	*-30.2*	*0.6*	*25.8*	*-0.5*	*-13.9*	*0.0*	*-2.2*	*5.2*	*-22.0*

Table 3-8. Differences in land use loading estimated (mass and percent) between the model form we present in this study and the standard Bayesian formulation using unadjusted land use map estimates. Negative values indicate lower predictions by the model form we present.

	Water		Urban		Sparse Development		Forest		Pasture & Hay		Crops		Wetlands		Residual	
Watershed	mass	%	mass	%	mass	%	mass	%	mass	%	mass	%	mass	%	mass	%
Ausable	-0.2	-9.4	0.5	18.3	1.1	14.8	-5.0	-16.7	0.1	18.5	-0.2	-17.4	-0.5	-90.8	3.1	-28
Bouquet	-0.1	-17.3	0.3	18	0.5	15.2	-2.5	-16.7	0.1	25.6	-0.5	-15.6	-0.3	-86.4	-1.7	-28.8
Great Chazy	-0.1	-16.1	0.3	20	0.7	13.7	-2.0	-16.5	0.4	28.1	-0.4	-13.1	-0.5	-126.9	-1.6	-30.5
Lamoille	-0.2	-12	1.3	17.4	1.6	14.9	-6.2	-16.6	1.2	26.3	-1.2	-14.9	-0.3	-106.1	-4.2	-27.7
LaPlatte	0.0	-31.8	0.1	16.6	0.1	13.3	-0.2	-14.2	0.3	29	-0.1	-13.3	0.0	0	-0.1	2.7
Lewis	0.0	3.2	0.0	5	0.1	11.9	-0.5	-15.8	0.3	29	-0.2	-16.2	-0.1	-65.9	-0.1	1.2
Little Ausable	0.1	0.0	0.1	18	0.2	14.3	-0.6	-17.2	0.2	32.8	-0.1	-10	0.0	0	-0.5	-15.2
Little Chazy	0.0	26.7	0.1	10.8	0.2	15.3	-0.4	-19	0.1	25.3	-0.2	-19.9	-0.2	-172	-0.3	-25.4
Little Otter	0.1	0.0	0.2	22.1	0.1	14.5	-0.3	-19.8	0.4	24.8	-0.2	-11.5	0.0	-14.9	0.1	-2.8
Mettawee	-0.1	-6.7	1.3	17.3	1.0	14.4	-2.6	-16.8	1.2	26.4	-0.6	-14	-0.6	-116.7	-2.4	-30.2
Missisquoi	-0.2	-11.5	2.2	17.6	1.4	14.9	-6.9	-16.5	1.7	25.5	-2.0	-14	-0.4	-97.5	-1.7	3.3
Otter	-0.1	-6.9	2.0	17.5	1.6	14.4	-6.4	-16.6	2.8	26.2	-1.4	-14.8	-1.2	-102.1	-2.8	17.1
Poultney	-0.2	-10.4	0.6	18	0.5	13.6	-1.9	-16.2	0.7	24.9	-0.3	-16.6	-0.5	-127.9	-0.8	16.4
Putnam	0.0	1.6	0.0	26.9	0.1	12.9	-0.5	-15.1	0.0	28.7	0.0	-0.5	-0.1	-145	-0.5	-18.9
Salmon	0.0	-11.9	0.1	27.7	0.1	11.4	-0.6	-17.1	0.1	35.5	0.0	-12.1	-0.1	-72.2	-0.5	-16
Saranac	-1.0	-10.2	0.7	18.1	1.4	14.6	-5.1	-16.4	0.2	25.1	-0.2	-21.6	-1.4	-102.6	-5.4	-18.5
Winooski	-0.2	-9.2	3.7	17.7	3.0	14.6	-9.4	-16.6	1.4	26.5	-1.2	-14.2	-0.3	-99.6	-3.9	6
Mean	*-0.1*	*1.8*	*0.8*	*18.1*	*0.8*	*14.0*	*-3.0*	*-16.7*	*0.7*	*27.0*	*-0.5*	*-14.1*	*-0.4*	*-84.8*	*-1.4*	*-11.5*

References

Allan, J. D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology Evolution and Systematics **35**:257-284.

Borah, D. K., G. Yagow, A. Saleh, P. L. Barnes, W. Rosenthal, E. C. Krug, and L. M. Hauck. 2006. Sediment and nutrient modeling for TMDL development and implementation. Transactions of the Asabe **49**:967-986.

Broad, S. T., and R. Corkrey. 2011. Estimating annual generation rates of total P and total N for different land uses in Tasmania, Australia. Journal of Environmental Management **92**:1609-1617.

Budd, L. F., and D. W. Meals. 1994. Lake Champlain Nonpoint Source Pollution Assessment. Technical Report No. 6A. Lake Champlain Management Conference, Grand Isle, VT.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen. Ecological Applications **8**:559-568.

Cha, Y., C. A. Stow, K. H. Reckhow, C. DeMarchi, and T. H. Johengen. 2010. Phosphorus load estimation in the Saginaw River, MI using a Bayesian hierarchical/multilevel model. Water Research **44**:3270-3282.

ESRI. 2012. ArcGIS 10.1. Environmental Systems Research Institute, Redlands, CA.

Ford, R. F. 2012. Agricultural Phosphorus in Vermont's Missisquoi Bay Watershed: History, Status, and Solutions. University of Vermont, Burlington, VT.

Fry, J. A., G. Xian, S. Jin, J. Dewitz, C. G. Homer, L. Yang, C. A. Barnes, N. D. Herold, and J. D. Wickham. 2011. Completion of the 2006 National Land Cover Database for the conterminous United States. Photogrammetric Engineering & Remote Sensing **77**.

Gelman, A., and J. Hill. 2009. Data Analysis Using Regression and Multilevel/Hierarchical Models. Cambridge University Press, New York, NY.

Gudimov, A., E. O'Connor, M. Dittrich, H. Jarjanazi, M. E. Palmer, E. Stainsby, J. G. Winter, J. D. Young, and G. B. Arhonditsis. 2012. Continuous Bayesian network for studying the causal links between phosphorus loading and plankton patterns in Lake Simcoe, Ontario, Canada. Environmental Science & Technology **46**:7283-7292.

Hegman, W., D. Wang, and C. Borer. 1999. Estimation of Lake Champlain Basinwide Nonpoint Source Phosphorus Export. Technical Report No.31. Lake Champlain Basin Program, Grand Isle, VT.

Hirsch, R. M., and L. A. De Cicco. 2014. User guide to Exploration and Graphics for RivEr Trends (EGRET) and dataRetrieval - R packages for hydrologic data (version 2.0, February, 2015).

Hirsch, R. M., D. L. Moyer, and S. A. Archfield. 2010. Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay river inputs. JAWRA Journal of the American Water Resources Association **46**:857-880.

Jin, S. M., L. M. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment **132**:159-175.

Johnes, P. J. 1996. Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. Journal of Hydrology **183**:323-349.

Kline, M., and B. Cahoon. 2008. Restoring Ecological Processes in Protected River Corridors. River Management Program, Vermont Agency of Natural Resources, Waterbury, VT.

Langendoen, E. J., A. Simon, L. Klimetz, N. Bankhead, and M. E. Ursic. 2012. Quantifying Sediment Loadings from Streambank Erosion in Selected Agricultural Watersheds Draining to Lake Champlain. Technical Report No. 72. Lake Champlain Basin Program, Grand Isle, VT.

LCBP. 2012. State of the Lake and Ecosystem Indicators Report. Lake Champlain Steering Committee, Lake Champlain Basin Program, Grand Isle, VT.

Liu, M., and J. Lu. 2013. Solution of export coefficients of nitrogen from different land-use patterns based on Bayesian analysis. Water Science and Technology **68**:632-640.

Medalie, L., R. M. Hirsch, and S. A. Archfield. 2012. Use of flow-normalization to evaluate nutrient concentration and flux changes in Lake Champlain tributaries, 1990–2009. Journal of Great Lakes Research.

Olofsson, P., G. M. Foody, S. V. Stehman, and C. E. Woodcock. 2013. Making better use of accuracy data in land change studies: Estimating accuracy and area and quantifying uncertainty using stratified estimation. Remote Sensing of Environment **129**:122-131.

Plummer, M. 2013. JAGS version 3.4.0 manual.

Plummer, M. 2014. rjags: Bayesian graphical models using MCMC. R package version 3-14. http://CRAN.R-project.org/package=rjags.

Preston, S. D., R. B. Alexander, G. E. Schwarz, and C. G. Crawford. 2011. Factors Affecting Stream Nutrient Loads: A Synthesis of Regional SPARROW Model Results for the Continental United States. JAWRA Journal of the American Water Resources Association **47**:891-915.

Qian, S. S., T. F. Cuffney, I. Alameddine, G. McMahon, and K. H. Reckhow. 2010. On the application of multilevel modeling in environmental and ecological studies. Ecology **91**:355-361.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Reckhow, K. H., M. N. Beaulac, and J. T. Simpson. 1980. Modeling phosphorus loading and lake response under uncertainty: A manual and compilation of export coefficients. Washington D.C.

Riggs, D. S., J. A. Guarnieri, and S. Addelman. 1978. Fitting straight lines when both variables are subject to error. Life Sciences **22**:1305-1360.

Shirmohammadi, A., I. Chaubey, R. D. Harmel, D. D. Bosch, R. Munoz-Carpena, C. Dharmasri, A. Sexton, M. Arabi, M. L. Wolfe, J. Frankenberger, C. Graff, and T. M. Sohrabi. 2006. Uncertainty in TMDL models. Transactions of the Asabe **49**:1033-1049.

Shrestha, S., F. Kazama, L. T. H. Newham, M. S. Babel, R. S. Clemente, H. Ishidaira, K. Nishida, and Y. Sakamoto. 2008. Catchment scale modelling of point source and non-point source pollution loads using pollutant export coefficients determined from long-term in-stream monitoring data. Journal of Hydro-Environment Research **2**:134-147.

Stehman, S. V. 2013. Estimating area from an accuracy assessment error matrix. Remote Sensing of Environment **132**:202-211.

Tetra Tech. 2013. Lake Champlain Basin SWAT Model Configuration, Calibration, and Validation. Prepared for U.S. EPA Region 1 - New England. Boston MA. Available from http://www.epa.gov/region1/eco/tmdl/lakechamplain.html.

Troy, A., D. Wang, D. Capen, J. O'Neil-Dunne, and S. MacFadden. 2007. Updating the Lake Champlain Basin Land Use Data to Improve Prediction of Phosphorus Loading. Technical Report No. 54. Lake Champlain Basin Program, Grand Isle, VT.

Vermont Center for Geographic Information. 2006. Vermont Hydrologically Corrected Digital Elevation Model.*in* VCGI, editor. VCGI, Waterbury, VT.

Wickham, H. 2009. ggplot2: elegant graphics for data analysis. Springer, New York.

Wickham, J. D., S. V. Stehman, J. A. Fry, J. H. Smith, and C. G. Homer. 2010. Thematic accuracy of the NLCD 2001 land cover for the conterminous United States. Remote Sensing of Environment **114**:1286-1296.

Wickham, J. D., S. V. Stehman, L. Gass, J. Dewitz, J. A. Fry, and T. G. Wade. 2013. Accuracy assessment of NLCD 2006 land cover and impervious surface. Remote Sensing of Environment **130**:294-304.

Zobrist, J., and P. Reichert. 2006. Bayesian estimation of export coefficients from diffuse and point sources in Swiss watersheds. Journal of Hydrology **329**:207-223.

CHAPTER 4. QUANTITATIVE DECISION AIDS FOR ILLUSTRATING TRADE-OFFS UNDER UNCERTAINTY IN WATERSHED MANAGEMENT FOR LAKE CHAMPLAIN

Introduction

The primary difficulty of making good decisions in multi-objective situations is that most of the time, getting more of everything good and less of everything bad is impossible. Alternative courses of action provide different levels of benefits and costs, so choosing one course over another means making compromises between objectives. The difficulty grows when the outcomes of those alternative courses of action – and therefore their benefits and costs – are uncertain. In these situations, where trade-offs are necessary and uncertainty is important, making good decisions requires a process that provides the best chances of a good outcome (Hammond et al. 1999). Ideally, that process should include attention to value-based objectives (Keeney 1992), the development of creative alternative courses of action (Gregory and Keeney 1994), and attention to the effects of uncertainty and risk (Gregory et al. 2012).

In the context of managing water quality in large catchments, decision making requires making value-based trade-offs, for example between cost and effectiveness, or between equitable distributions of financial burden and spatially targeted management actions. Many sources of uncertainty – about the effectiveness of management practices, about the connections between those effects and relevant environmental targets, and about the role of natural variability – all add to the difficulty in choosing management strategies. In the face of these difficult conditions, a culture of increased public scrutiny has led to a desire for increased scientific rigor in analyses, in the hopes that more and better science will produce better decisions. (Gregory et al. 2006, Hirsch et al. 2006).

One function of the US Environmental Protection Agency's Total Maximum Daily Load (TMDL) program is to provide a rigorous, scientific, and consistent basis for setting water quality management targets (EPA 1999). To enable this level of rigor, watershed models that represent the important hydrologic and sediment-related processes are used to predict the consequences of management actions on pollutant loads and waterbody responses (NRC 2001). However, for large watersheds, these models quickly become very complex, and thus don't deal effectively with uncertainty (Shirmohammadi et al. 2006, Radcliffe et al. 2009). In addition, TMDL analyses do not often consider management costs, distribution of burden or other value-based considerations that are likely to guide the implementation of management plans (EPA 1999), and therefore do not illustrate important trade-offs that water resource managers will face.

To enable good watershed management decisions, resource managers need models that provide insight into the difficult parts of those decisions – the role of widespread uncertainty and variability, and the consequences of value-based trade-offs between objectives.

A variety of tools exist for rigorously addressing uncertainty and for helping to illustrate trade-offs clearly. Decision analysis provides a framework for integrating those tools and others to support good decision making. Bayesian networks (BNs) have gained widespread use recently in the area of natural resources management because of their ability to integrate different sources of information, to connect decision points to management objectives clearly and intuitively, and to rigorously quantify and propagate uncertainty through the connections in the model. They express uncertain quantities in the model as marginal probability distributions. These distributions are connected to other uncertain nodes in the model through functional or correlational relationships, which can then be quantified as conditional, or joint probabilities. Uncertainty is propagated through the network via these joint probability distributions (Jensen and Nielsen 2007). This form of model has been used extensively in water resources management to aid decisions in a wide variety of contexts, including for groundwater management (Farmani et al. 2012), for flow-regime restoration (Said 2006), for fisheries and river rehabilitation (Varis and Kuikka 1999, Borsuk et al. 2012), for coastal eutrophication management (Borsuk et al. 2004, Barton et al. 2006), and for freshwater eutrophication management in lakes and in rivers (Ames et al. 2005, Gudimov et al. 2012).

Optimization is a method for assessing trade-offs quantitatively and can be used in conjunction with water quality modeling for water resources planning (Williams 1996, Cerucci and Conrad 2003, Muleta and Nicklow 2005, Gitau et al. 2006, Martin et al. 2011, Rodriguez et al. 2011). It is particularly useful in situations where complicated relationships between cost and effectiveness exist due to non-linear or discontinuous cost-effectiveness functions or complex spatial relationships.

Evolutionary, or genetic, optimization is particularly useful for very complex problems, and is based on a stochastic approach. In this method an initial random population of solutions is selected and then evaluated relative to a fitness function. To produce a second generation of solutions, the algorithm keeps those that are the “fittest,” but also generates new solutions through a process of mutations and crossovers of existing solutions. At each generation, the best solutions are maintained and passed into subsequent generations. Through many repeated iterations of the programs, evolutionary algorithms can define a set of optimal solutions that provide equal return across the objectives, referred to as the efficiency frontier.

The primary goal of this study was to demonstrate an approach to help TMDL decision-makers address two questions:

1) what is the likelihood that a particular management strategy will result in compliance with watershed loading targets, and

2) what is magnitude of the trade-offs between competing management approaches?

To do this, we developed a Bayes network (BN) to predict phosphorus loads in four major tributaries to Lake Champlain in Vermont, USA, that accounts for uncertainty and variability in land-use specific phosphorus loading rates, uncertainty and variability in the effectiveness of twenty common phosphorus management practices, and variability in the cost to implement those practices. Given a user-defined scenario, the BN estimates the probability of attaining watershed-specific targets set under a TMDL, and the expected cost to implement that scenario. We then used evolutionary optimization within the context of the BNs to find the most efficient management portfolios under conditions of uncertainty. Lastly, we evaluated a selection of these portfolios along the efficiency frontier against competing management objectives in a multiple-criteria decision analysis framework (Kirkwood 1992) to demonstrate the trade-offs that exist between probability of compliance, expected cost of management, and equity in the distribution of management burden between land use sectors.

Methods

Site Description

Lake Champlain is a large lake that lies between the US states of Vermont and New York, and the Canadian province of Quebec (Figure 4-1). Some areas of the lake experience extensive blue-green algae blooms for large portions of the growing season, likely a substantial result of excessive non-point source phosphorus loading from tributaries. The states of Vermont and New York initially developed a phosphorus TMDL for Lake Champlain in 2001, and the Vermont portion is currently under revision. Most management efforts in the basin are aimed at reducing the capacity of developed and agricultural land uses to export phosphorus from the landscape, with the intent to reduce end-of-tributary phosphorus loads and then in-lake phosphorus concentrations.

We selected for inclusion in this study the four largest Vermont tributaries, which collectively represent 74% of the land area in the Vermont portion of the Lake Champlain Basin. We opted to test our approach on more than one watershed because the relative proportion of anthropogenic land uses differs strongly between them (Table 4-1). Cost-efficient combinations of management actions are, therefore, likely to differ as well, and key trade-offs among objectives related to management strategies are likely to reflect these different characteristics. Understandably, the state of Vermont is only responsible for managing the phosphorus load coming from the portion of the watershed under its jurisdiction – therefore, all estimates of land use areas, phosphorus loads, and proportion of the manageable area in the Missisquoi watershed refer only to the portion of the watershed in the US.

Table 4-1. Land use areas and proportions for each watershed included in the modeling. Watersheds reading left to right in the table are organized north to south in the basin.

Land Uses	Missisquoi		Lamoille		Winooski		Otter	
	ha	%	ha	%	ha	%	ha	%
Urban	5514	2.5	3344	1.8	9282	3.3	5033	2.2
Sparse Development	9890	4.5	11297	6.0	22344	7.9	12341	5.5
Pasture & Hay	22876	10.4	14650	7.8	17258	6.1	34992	15.6
Cropland	13357	6.1	7642	4.1	7782	2.7	8876	4.0
Forest	163652	74.3	146728	78.0	222241	78.4	151869	67.8
Wetland	2958	1.3	2315	1.2	2242	0.8	9078	4.1
Water	2029	0.9	2037	1.1	2455	0.9	1830	0.8
Total	220276	-	188014	-	283604	-	224019	-

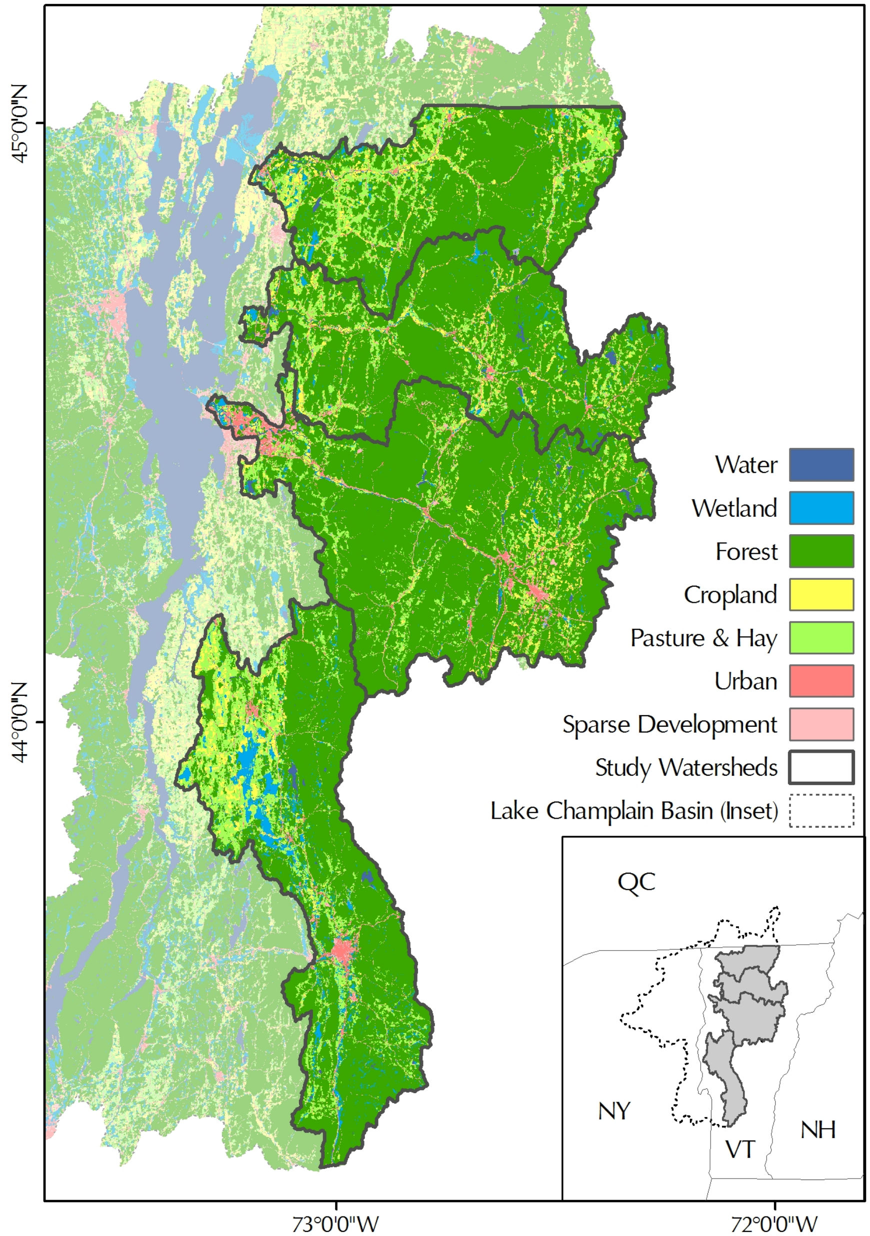

Figure 4-1. Location and land use of the watersheds included in this study.

Model Development

We developed an influence diagram to describe the basic conceptual structure of our model (Figure 4-2). In our model, land management actions reduce the average phosphorus export rate of a land use, which in turn reduces the tributary phosphorus loads. Our model is not intended to replicate watershed processes; instead, connections in Figure 4-2 are meant to signify probabilistic or correlational relationships. Ellipses represent quantities subject to random variation or uncertainty, rectangles signify decision points, and hexagons represent value-based objectives that we infer based on stakeholder concerns, and the goals of the TMDL program (EPA 1999, VT DEC 2014). However, because the State's TMDL implementation program addresses only tributary phosphorus loading with an assumed relationship to in-lake concentrations, we evaluated the effects of management scenarios only on objectives related to the watershed processes portion (left side) of the diagram.

We implemented the model in Analytica, a software for evaluating object-oriented graphical probability models through Monte Carlo or Latin Hypercube sampling (Analytica 2012).

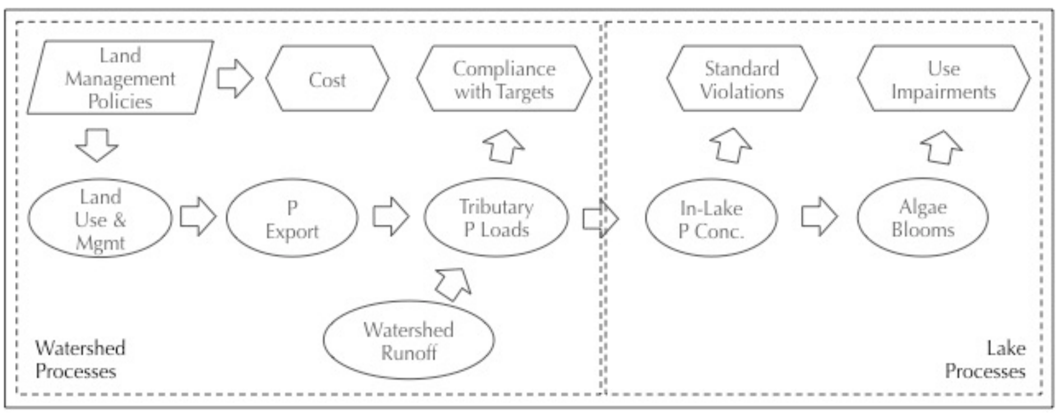

Figure 4-2. Conceptual model linking land management policy, land use, tributary phosphorus loads, and algae blooms. The model described in this study concerns the left side of the diagram only.

For purposes of this modeling effort, we chose to include management strategies that are currently in common use in the Lake Champlain Basin, or that have recently been proposed by the State of Vermont as part of their draft implementation plan (State of Vermont 2014). A list of these strategies and the land uses they impact are given in Table 4-2. Our intention was not to include every possible management practice that can reduce phosphorus loading, but instead to include the most widely applicable strategies that collectively are likely account for the vast majority of management action.

Reduction efficiencies associated with each management practice were compiled from a literature review. To the extent possible, we only included results from areas with similar environmental conditions to the Lake Champlain Basin (i.e., cool, humid, temperate). For some agricultural practices, however, only data from other areas was available. In these cases, we only included efficiencies from cropping systems that were similar (e.g., from silage corn fertilized with dairy manure). Reduction efficiencies reported in the literature were recorded as percentage reductions, and were combined into triangular distributions by sampling with equal weighting from each of the efficiencies reported. Therefore, the variance associated with the reduction efficiencies reflects a combination of parameter uncertainty around the "true" reduction rates and natural variability in effectiveness across time and location.

Table 4-2. Land management practices included in the model. Upper bounds on management practice values are given in Appendix C.

Land Use	Possible Management Practices	Range of Possible Values
Urban Areas (designated MS4)	Stormwater runoff treatment from roads	0 to # of acres of impervious surfaces
	BMP design size for roads	0.25”, 0.5”, 0.9”, 1.5”, or 2.0” storm design size
	Stormwater runoff treatment from “other” impervious	0 to # of acres of impervious surfaces
	BMP design size for “other” impervious	0.25”, 0.5”, 0.9”, 1.5”, or 2.0” storm design size
Sparse Development (outside MS4 designation)	Stormwater runoff treatment from roads	0 to # of acres of impervious surfaces
	BMP design size for roads	0.25”, 0.5”, 0.9”, 1.5”, or 2.0” storm design size
	Stormwater runoff treatment from “other” impervious	0 to # of acres of impervious surfaces
	BMP design size for “other” impervious	0.25”, 0.5”, 0.9”, 1.5”, or 2.0” storm design size
Cropland	Cover Cropping	0 to # of acres of available cropland
	Reduced Tillage	0 to # of acres of available cropland
	Manure Injection	0 to # of acres of available cropland
	Lowering Erosion Standard for SFOs	0 or 1 (affects all farms equally)
	Management of Tile Drainage Water	0 to # of acres of tile-drained cropland
	Riparian Buffer Installation	0 to # of miles of streams with no buffer
	Diversion of runoff from SFO barnyards	0 to # of farms
Pasture & Hay	Livestock Exclusion from streams	0 to # of acres of pasture
	Manure Injection on hayland	0 to # of acres of hayland
Wetlands	Cropland to Wetland Conversion	0 to # of acres of current cropland overlying wetland areas
	Hayland to Wetland Conversion	0 to # of acres of current hayland overlying wetland areas
Forest	Increased backroad maintenance	0 to # of miles of dirt road

Cost data were compiled from local resource management agencies, and reflect all direct costs related to the implementation of management practices, such as the cost of building structures (detention ponds, barnyard runoff diversion structures, etc.) and payments to land owners for voluntary programs (e.g, state cover cropping programs), but do not include costs associated with operation and maintenance of structures, program administration, or land purchases necessary to install practices. We assumed cost increased linearly with increased implementation. All costs are annualized over 20 years at 2% interest to enable fair comparisons between long-lived stormwater management practices and field-based agricultural practices that are renewed every year. Cost estimates for each practice on a per hectare basis were complied into triangular distributions using a similar method as for the effectiveness data. However, the variance of these distributions reflects variability in the costs to implement practices rather than uncertainty about any true cost of implementation.

Land Use Data & Phosphorus Loading Rates

Using the National Land Cover Database (NLCD) 2006 product as a base map (Fry et al. 2011), we developed bias-adjusted estimates of land use areas for each watershed. Wickham et al. (2013) conducted an accuracy assessment for this dataset, estimating rates of errors of omission and of commission relative to the reference imagery. Following methods described in Stehman (2013), we used the data from this accuracy assessment to add errors of omission to the map estimate of each land use area while subtracting errors of commission to yield the true area of each land use.

We then used these land use areas together with total phosphorus loading data from the years 2006-2013 to estimate land use-specific phosphorus loading rates for 17 watersheds throughout the Lake Champlain Basin. We applied a Bayesian hierarchal framework to the standard export coefficient model (Reckhow et al. 1980), which yielded phosphorus loading rate estimates that incorporated inter-annual hydrologic variability (from the loading data) and uncertainty in land use classification, along with estimates of the watershed-scale residual loading. We used the estimates for these parameters together with their uncertainty and the total model error to estimate baseline total watershed phosphorus loads. A fuller description of the methods used to estimate the bias-adjusted land use areas, phosphorus loads, and land use loading rates is given in Chapter 3 of this dissertation.

These data allowed us to estimate the probability density functions for the total phosphorus load for each watershed and to calculate the probability that the load in any given year would fall below the watershed target identified in the new TMDL analysis. As part of this analysis, EPA developed non-point source loading targets for each lake segment watershed. Because the watersheds we included in this study are within those lake segment watershed areas, we took as our targets a proportion of the NPS allocation equal to the ratio between our watersheds and the lake segment watersheds.

Estimating Reductions

To estimate reductions resulting from the implementation of Best Management Practices (BMPs), we multiplied the baseline loading rates by the product of the proportion of land available for a particular practice and the reduction efficiency for that practice in the following manner:

$$MLR_i = ULR_i * \left[1 - \left(\frac{BMP_{1\,-\,treated}}{BMP_{1\,-\,possible}} * E_1\right)\right] * \cdots * \left[1 - \left(\frac{BMP_{j\,-\,treated}}{BMP_{j\,-\,possible}} * E_j\right)\right]$$

where MLR_i is the managed loading rate for land use *i*, ULR_i is the baseline loading rate for the same land use, BMP_j is a particular BMP, and E_j is the reduction efficiency of BMP_j.

This provided an area-weighted average loading rate after BMP implementation (MLR, above). This method assumes that all BMPs within a particular land use can be applied to the same hectare of land. In our case, we chose BMPs that are considered in our area to be stackable in this way.

Optimization Methods

We employed the standard evolutionary optimization engine within Analytica (built by Frontline Solutions) to generate solutions that sought to minimize the expected value of cost of management within each watershed while maximizing the probability of compliance with the watershed loading targets. We terminated the search once the algorithm was unable to find any improved solutions for 60 seconds. We ran the optimizer engine enough times to generate 100,000 unique solutions per watershed to help define the efficiency frontier between those two objectives. We then analyzed those solutions in a number of ways, as described below.

Results and Discussion

Probabilities associated with values for the total phosphorus load under the baseline management scenario are shown in Figure 4-3 (solid lines). Median values for the current total load were 110.7 metric tons (mt) in the Missisquoi, 54.7 mt in the Lamoille, 100.8 mt in Otter Creek, and 179.5 mt in the Winooski, which represent 5.6%, 20.8%, 18.8%, and 28.9% probabilities of compliance, respectively, with the watershed loading targets under the new TMDL. Confidence limits for the watershed loads and land use sector loads are given in Table 4-3.

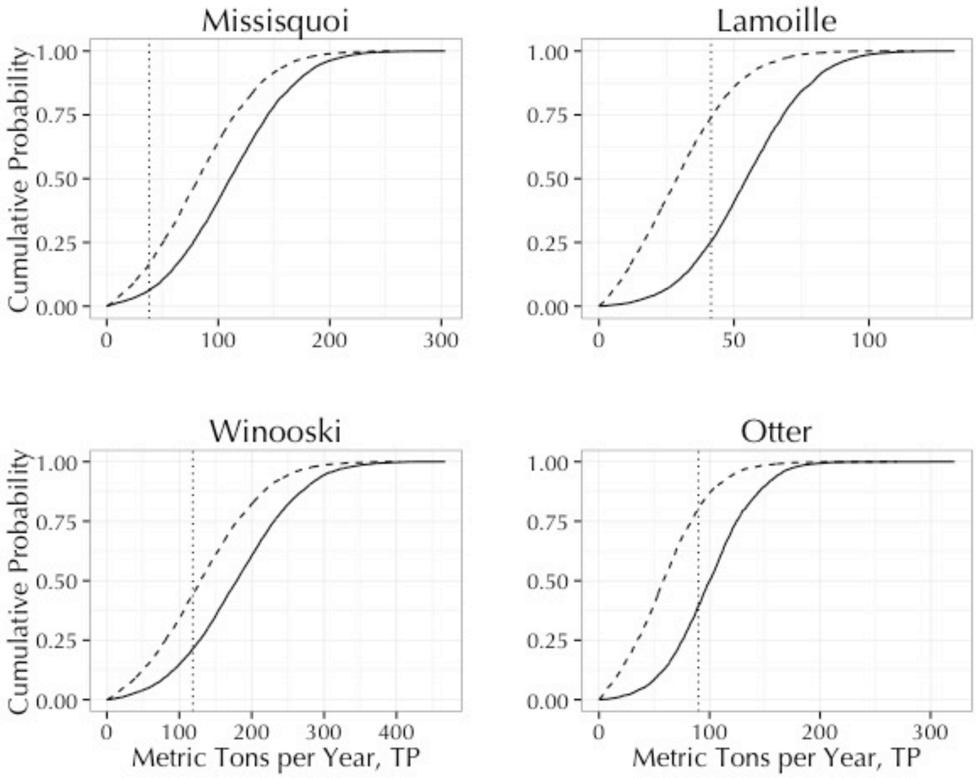

Figure 4-3. Cumulative distribution functions for total phosphorus load under the baseline scenario (solid lines) and the "do everything everywhere" scenario (dashed lines). Vertical dotted lines are watershed-specific NPS loading targets.

In contrast to the current management scenario, where relatively few of the non-point phosphorus sources have received attention, we evaluated a “do everything everywhere” scenario, where each management practice is implemented at the fullest possible extent. The effects of this scenario varied widely between watersheds. For example, end-of-tributary phosphorus reductions ranged from 34.6% in the Missisquoi to 89.7% in the Lamoille. This scenario also increased the probability of compliance in every watershed, but to varying degrees (Figure 4-3, dashed lines, Table 4-4). In the Missisquoi, for example, full-scale management increased the probability of compliance to 15.6%, an improvement of 10% over the baseline scenario, whereas in the Lamoille, the probability of compliance was raised to 69%, an improvement of 48%.

Table 4-3. Median baseline total phosphorus loads for each land use sector within each watershed, with credible intervals (CI) and probability that the watershed is in compliance with its loading target.

Watershed	Land Use Sectors	Median Baseline Load (mt/yr)	Lower 90% CL Load	Upper 90% CL Load	Probability of Compliance
Missisquoi	Developed Lands	11.3	5.6	17.5	
	MS4	0.0	0.0	0.0	
	Non-MS4	11.3	5.6	17.5	
	Agriculture	19.2	8.1	32.6	
	Hay & Pasture	8.9	1.5	18.5	
	Crops	9.8	1.5	21.0	
	Forest	35.2	12.0	59.2	
	Wetlands	0.3	-0.9	1.5	
	Water	0.6	-0.6	1.9	
	Total Watershed	*110.7*	*32.5*	*193.0*	*5.6*
Lamoille	Developed Lands	19.3	9.2	30.7	
	MS4	1.0	0.4	1.5	
	Non-MS4	18.3	8.7	29.2	
	Agriculture	13.6	5.7	23.5	
	Hay & Pasture	5.5	0.9	11.4	
	Crops	7.9	1.2	16.9	
	Forest	47.3	16.1	79.6	
	Wetlands	0.2	-0.5	0.9	
	Water	1.7	-1.7	5.1	
	Total Watershed	*54.7*	*20.8*	*88.4*	*20.8*
Winooski	Developed Lands	45.1	22.2	70.4	
	MS4	9.2	3.9	14.8	
	Non-MS4	35.9	17.1	57.0	
	Agriculture	14.8	6.3	25.3	
	Hay & Pasture	6.4	1.1	13.4	
	Crops	8.0	1.2	17.2	
	Forest	71.7	24.4	120.6	
	Wetlands	0.2	-0.5	0.9	
	Water	2.0	-2.1	6.1	
	Total Watershed	*179.5*	*57.0*	*309.2*	*18.8*
Otter Creek	Developed Lands	24.7	12.1	38.6	
	MS4	1.3	0.5	2.2	
	Non-MS4	23.3	11.4	36.6	
	Agriculture	22.6	9.4	38.9	
	Hay & Pasture	13.0	2.2	27.2	
	Crops	9.2	1.4	19.6	
	Forest	49.0	16.7	82.4	
	Wetlands	0.7	-2.0	3.5	
	Water	1.5	-1.5	4.6	
	Total Watershed	*100.8*	*36.7*	*163.7*	*28.9*

Table 4-4. Expected P reductions, cost, post-management load, and probability of compliance resulting from the "do everything everywhere" management scenario.

Watershed	Land Use Sectors	P Reductions (mt/yr)	P Reductions (%)	Expected Cost ($M/yr)	Median Post-Management Load (mt/yr)	Probability of Compliance
Missisquoi	Developed Lands	10.9	96.6	20.2	0.4	
	MS4	0.0	0.0	0.0	0.0	
	Non-MS4	10.9	96.6	20.2	0.4	
	Agriculture	17.4	90.7	7.0	1.8	
	Hay & Pasture	7.6	85.9	2.2	1.3	
	Crops	9.4	95.9	4.8	0.4	
	Forest	0.4	1.2	2.4	34.8	
	Wetlands	-0.3	-94.3	2.3	0.6	
	Total Watershed	*29.9*	*34.6*	*32.0*	*82.2*	*15.68*
Lamoille	Developed Lands	18.6	96.6	28.5	0.7	
	MS4	0.9	96.8	1.6	0.0	
	Non-MS4	17.7	96.6	26.9	0.6	
	Agriculture	12.5	91.4	4.0	1.2	
	Hay & Pasture	4.7	85.8	1.4	0.8	
	Crops	7.6	96.0	2.6	0.3	
	Forest	0.5	1.1	3.3	46.8	
	Wetlands	-0.2	-117.2	1.7	0.4	
	Total Watershed	*26.4*	*89.7*	*37.5*	*28.8*	*69.0*
Winooski	Developed Lands	43.5	96.5	73.4	1.6	
	MS4	8.9	96.8	20.1	0.3	
	Non-MS4	34.7	96.6	53.3	1.2	
	Agriculture	13.4	90.7	5.3	1.4	
	Hay & Pasture	5.5	85.7	1.5	0.9	
	Crops	7.7	95.6	3.8	0.4	
	Forest	0.9	1.3	5.2	70.8	
	Wetlands	-0.2	-101.8	1.4	0.4	
	Total Watershed	*50.3*	*38.1*	*85.2*	*130.0*	*39.3*
Otter Creek	Developed Lands	23.8	96.6	36.5	0.8	
	MS4	1.3	96.8	2.5	0.0	
	Non-MS4	22.6	96.6	33.9	0.8	
	Agriculture	20.6	91.0	7.1	2.0	
	Hay & Pasture	11.5	87.9	3.1	1.6	
	Crops	8.8	96.4	4.0	0.3	
	Forest	0.4	0.8	2.7	48.6	
	Wetlands	-0.8	-111.2	6.2	1.5	
	Total Watershed	*43.0*	*69.6*	*52.5*	*59.4*	*72.8*

Cost to achieve these levels of compliance varied widely between watersheds. In the Missisquoi watershed, which has a relatively large proportion of agricultural land, the annualized cost of the full-scale management scenario was just under $35 million per year. In the Winooski, however, which has a larger proportion of developed land uses, the annualized costs were over $89 million per year. These differences are driven primarily by the much higher costs associated with the largest stormwater BMP design sizes used to address impervious area in the developed land areas. By using design sizes that are more cost efficient (0.9" storm sizes versus for 2.0" storm sizes), savings of 51% per year can be realized, with reductions in probability of compliance of between 0.5% and 3% (Table 4-5). This disproportionate effect of storm design size points to the existence of significant trade-offs between cost of a chosen management scenario and probability of compliance.

Table 4-5. Differences in P reductions, expected cost, and probability of compliance resulting from using a stormwater design size for 2.0" storms versus for 0.9" storms on all impervious area in each watershed.

Watershed	Stormwater BMP design size	P Reductions (%)	Expected Cost ($Millions/yr)	Probability of Compliance
Missisquoi	2.0" design size	96.7	20.2	15.6
	0.9" design size	85.7	9.1	15.2
Lamoille	2.0" design size	96.6	28.5	69.0
	0.9" design size	85.7	12.8	66.6
Winooski	2.0" design size	96.6	73.4	39.3
	0.9" design size	85.7	33.0	37.2
Otter Creek	2.0" design size	96.6	36.5	72.8
	0.9" design size	85.8	16.4	70.1

Collectively, results from the optimizer engine were able to delineate the relationship between cost and probability of compliance in each watershed. The efficiency frontier, along which the most cost-effective solutions lie, shows that for lower levels of management, large gains in probability of compliance can come from modest increases in the level of management. However, after a point, further raising the probability of compliance incurs steep increases in cost (Figure 4-4).

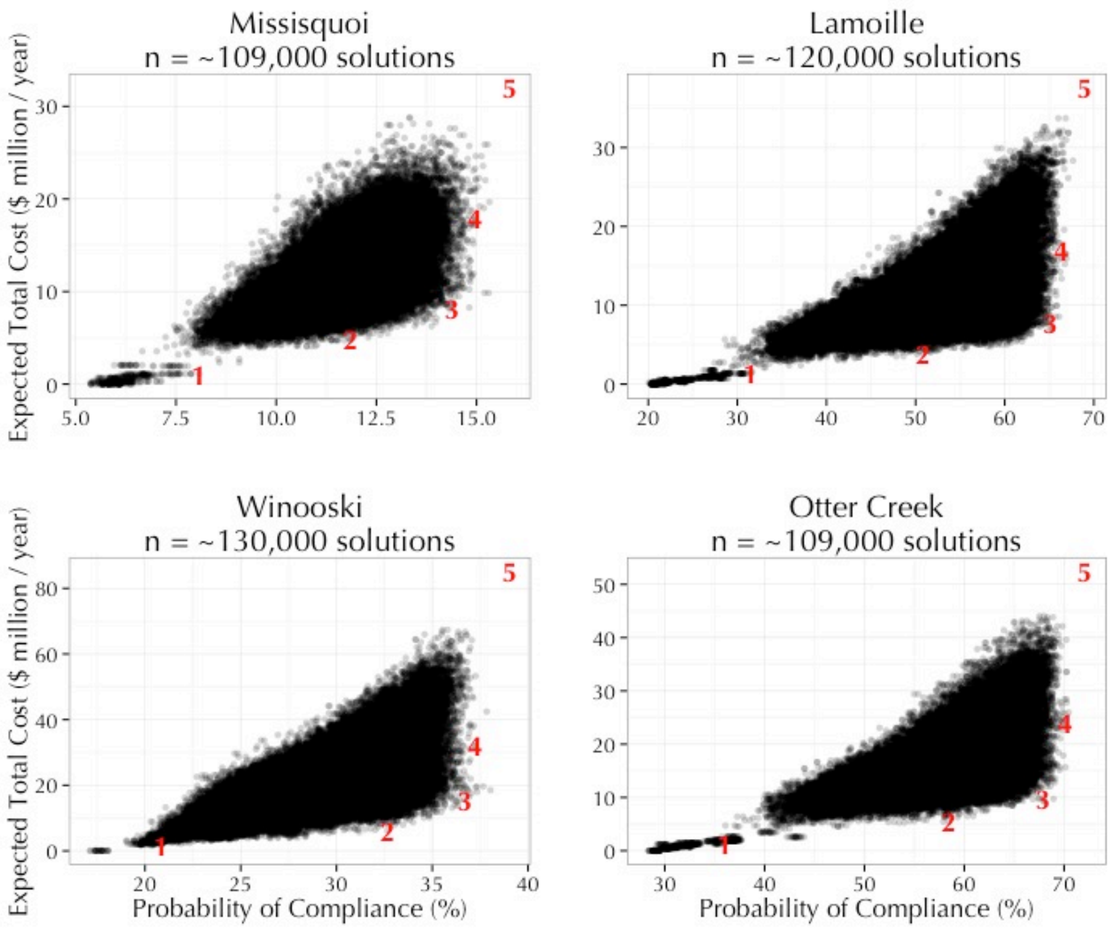

Figure 4-4. Probability of compliance and expected cost from ~100,000 management portfolios tested by the genetic algorithm (black dots), describing the efficiency frontier in each watershed. Scenarios along the efficiency frontier chosen for further analysis are designated by red numbers.

We selected five scenarios from points along the efficiency frontier (red numbers in figure 4) to illustrate the scale of trade-offs between three objectives: expected cost to implement the scenario, probability of compliance, and equity in the distribution of the burden associated with implementing the scenario between the developed and agricultural land uses. For this last objective, we defined and quantified two versions of equity – the first in terms of proportion of the total cost burden, and the second in terms of proportion of reductions, both relative to the proportion of the total load each land use sector contributions. For these evaluations, we created a coefficient that varied from -1 to +1. A value of 0 indicates that developed lands and agricultural lands are responsible for a proportion of the burden equal to their contribution to the total load. A value of -1 indicates that developed lands are responsible for the entire burden but contribute nothing to the problem. A value of +1 indicates a similarly unfair burden for agricultural land uses.

The consequences of each scenario in each watershed are summarized in Table 4-6 through Table 4-9. The pattern within objectives is similar in each watershed – the probability of compliance increases steeply between scenarios 1 (lowest cost and lowest probability of compliance) through 3, and more slowly between scenarios 3 through 5 ("do everything everywhere"). Cost, on the other hand, increases slowly between the first three scenarios, and much more quickly between the last three. Both equity measures show that at levels of spending below $15 million per year, agriculture bears much of the burden in load reduction and cost. At higher levels of spending, developed lands take more of the cumulative financial burden because of the much higher costs associated with those management practices, but the load reduction burdens become more equitable (Figure 4-5).

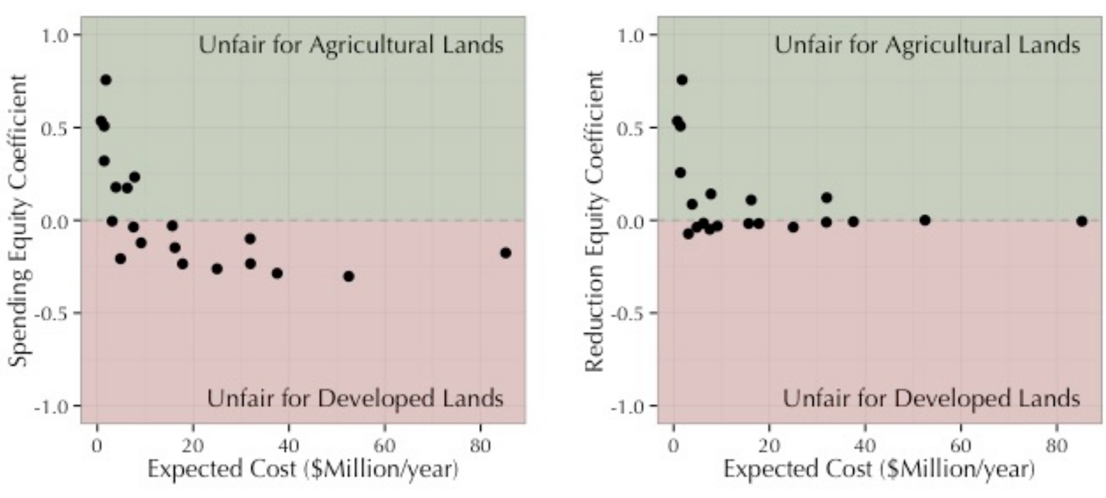

Figure 4-5. Trade-offs between cost and equity in terms of spending (left) and in terms of reductions (right). Results are shown for all scenarios in all watersheds.

Table 4-6. Consequences table for five management scenarios in the Missisquoi watershed.

Objective	Measure	Scenario M1	Scenario M2	Scenario M3	Scenario M4	Scenario M5
Prob. Of Compliance	%	7.8	11.3	14.2	14.3	15.7
Expected Total Cost	$M/yr	1.11	4.81	8.25	17.87	33.58
Equity in Spending	Coefficient	0.508	0.201	0.214	-0.186	-0.192
Equity in Reductions	Coefficient	0.508	0.057	0.094	0.130	0.124

Table 4-7. Consequences table for five management scenarios in the Lamoille watershed.

Objective	Measure	Scenario L1	Scenario L2	Scenario L3	Scenario L4	Scenario L5
Prob. Of Compliance	%	30.1	53.5	65.2	66.5	69
Expected Total Cost	$M/yr	1.42	3.09	7.54	17.79	37.52
Equity in Spending	Coefficient	0.320	-0.005	-0.036	-0.235	-0.286
Equity in Reductions	Coefficient	0.257	-0.072	-0.049	-0.018	-0.008

Table 4-8. Consequences table for five management scenarios in the Winooski watershed.

Objective	Measure	Scenario W1	Scenario W2	Scenario W3	Scenario W4	Scenario W5
Prob. Of Compliance	%	19.9	32.6	37.3	37.9	39.3
Expected Total Cost	$M/yr	1.78	6.23	15.65	31.89	85.23
Equity in Spending	Coefficient	0.757	0.174	-0.030	-0.100	-0.176
Equity in Reductions	Coefficient	0.757	-0.016	-0.017	-0.010	-0.005

Table 4-9. Consequences table for five management scenarios in the Otter Creek Watershed.

Objective	Measure	Scenario OC1	Scenario OC2	Scenario OC3	Scenario OC4	Scenario OC5
Prob. Of Compliance	%	35.1	57.0	67.9	69.2	71.9
Expected Total Cost	$M/yr	1.36	5.45	9.89	23.99	54.17
Equity in Spending	Coefficient	0.535	0.145	0.052	-0.175	-0.271
Equity in Reductions	Coefficient	0.535	-0.175	-0.039	-0.023	-0.001

Because no single scenario is unequivocally superior to all others, we used a basic utility function to describe the best overall performance of each scenario in each watershed, consistent with established techniques in decision analysis (Goodwin and Wright 2004). We assigned an importance weight to each objective, and then multiplied that weight by the scaled outcomes of each scenario. The sum of these weighted and scaled outcomes across objectives represents the overall performance of each scenario. However, because these weights should reflect the values of the decision-makers – in this case, the State of Vermont – we used a range of weights to demonstrate the effects of different sets of values on choosing a single scenario for implementation. We varied the weights on probability of compliance and expected cost one at a time, and in each case, split the remaining weight among the other two objectives (we considered equity to be one objective with two parts; thus the combined importance weights for both definitions of equity was equal to the weight for probability of compliance or expected cost).

Applying this approach to evaluating the scenarios for each watershed shows that under a range of importance weighting schemes, scenario 3 provides the best return across all objectives in each watershed (Figure 4-6). Only at very high weights on probability of compliance (i.e., when cost and equity don't carry much importance), the "do everything everywhere" scenario was preferred; in most cases, scenario 3 provided the best return across objectives (Figure 4-6, top). The same is true over a wide range of weights on expected cost (Figure 4-6, bottom).

These results suggest that striving for the highest levels of compliance ignores the importance of cost and of equitable distribution of the management burden, both of which are important stakeholder values (VT DEC 2014).

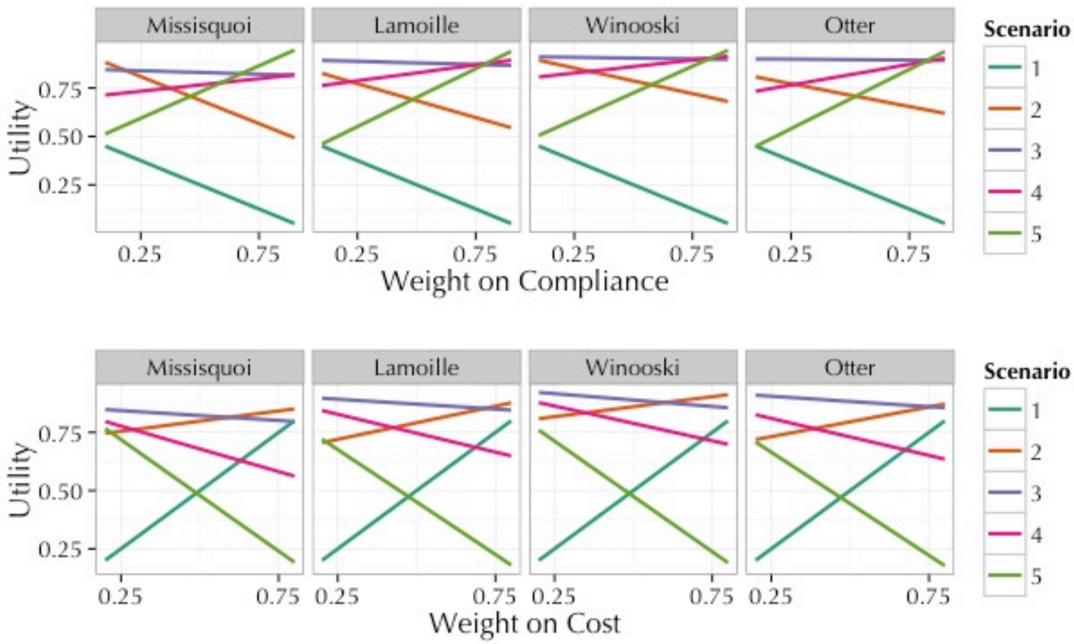

Figure 4-6. Overall utility for each example scenario, across ranges of weights on probability of compliance (top), and expected cost (bottom).

In our model, the probabilities of compliance with loading targets in each watershed are driven by two main factors specific to that watershed. The first is the gap between the current load and the loading target. EPA based the TMDL target loads on the results of their in-lake modeling which established the level of phosphorus load that each area of the lake could accommodate in order to achieve the in-lake concentration standards (Tetra Tech 2013). Differences in depth, internal loading rates, and hydraulic residence time (among other physiographic features) of the segments mean that some areas of the lake can accommodate much less phosphorus than other segments. The Missisquoi watershed, which drains to a wide and very shallow bay with low hydraulic flow rates out of the bay (Smeltzer 1997), has a low capacity for assimilating excess phosphorus, and therefore a relatively low loading target. However, the Missisquoi watershed is large and heavily agricultural and has a current average phosphorus load of more than twice the watershed target. The gap between the current load and the target will require phosphorus reductions of more than 65%, the largest challenge among the four watersheds we examined.

The second limiting factor in achieving high levels of compliance with watershed targets is the unmanageable non-point source phosphorus in the form of loading from forested lands, water, wetlands, and residual loading unexplained by land use. For the Missisquoi and Winooski watersheds, this load is substantial. While our estimates suggest that loading from water and wetlands is negligible (Table 4-3), it appears that residual loads (Figure 4-7, solid lines) and total unmanageable loads (Figure 4-7, dashed lines) are substantial, especially in relation to the watershed targets (dotted vertical line). The total unmanageable load can exceed the target in all four of our test watersheds. In each of the Missisquoi and the Winooski watersheds, the probability that this load exceeds the loading target is large - 95% and 60%, respectively. Therefore, our results suggest that even extremely high levels of management are unable to provide high levels of confidence that the phosphorus loads will comply with targets.

To make progress in improving water quality in impaired waters, managers must simultaneously keep the problem from getting worse in new areas while reducing impacts that already exist – that is, employ both conservation and restoration activities. This is a particularly important distinction for the developed land uses, as urbanization and development grows in Vermont. Though under revision, the current Vermont stormwater management manual calls for all new development to reduce new phosphorus loads by 40%. The modeling approach described here accounts for phosphorus reductions associated with stormwater retrofits (i.e., managing existing phosphorus loading from stormwater) but cannot account for the increased load from newly developed areas. To raise the probability of compliance with the watershed targets, efforts such as are described here to reduce existing phosphorus load will need to be paired with an additional set of tools, funding, and effort to reduce new phosphorus loads resulting from land use changes.

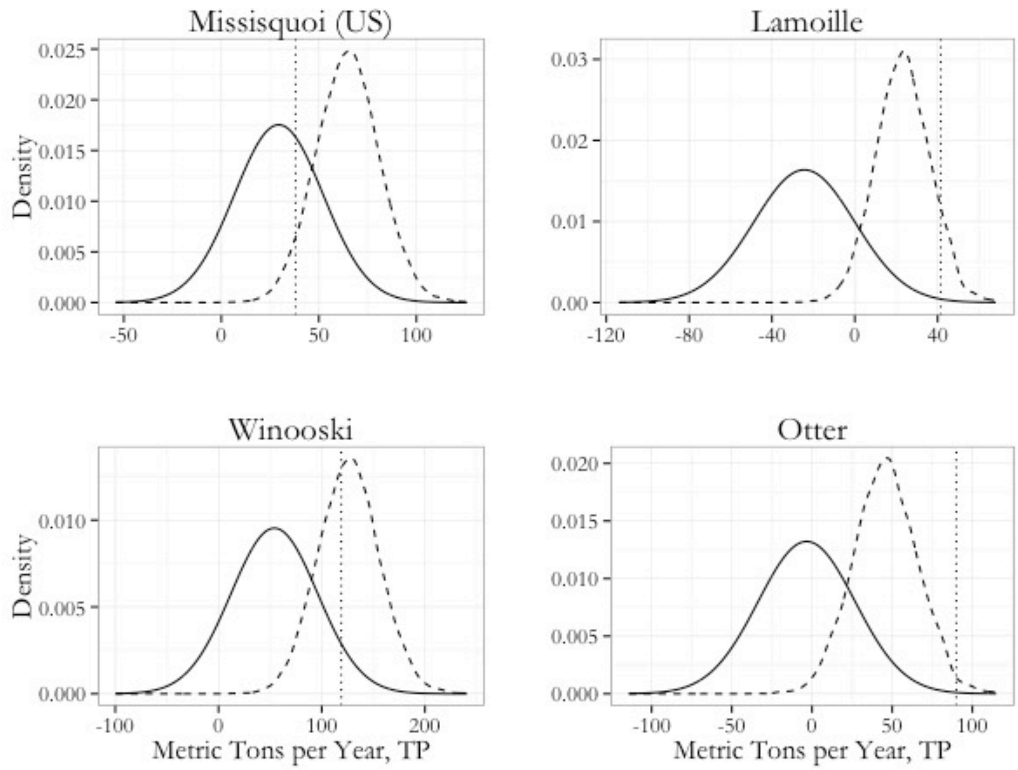

Figure 4-7. Probability density functions for residual loads (solid lines) and total unmanageable loads (residual plus forest loads, dashed lines) in relation to the watersheds targets (vertical, dotted lines).

Based on the trade-offs analyses discussed above, it is clear that non-linear trade-offs exist between probability of compliance, cost of management, and equity in the distribution of costs. In scenarios that approach the "do everything everywhere" option, an increase in compliance of 1 out of 100 years can come at a cost in the tens of millions of dollars per year. Whether that tradeoff is meaningful or acceptable is a value-based question that demands careful consideration.

Many previous implementations of Bayesian networks in watershed management contexts have demonstrated the value of understanding the role of uncertainty in the individual attainment of economic and environmental management objectives (Borsuk et al. 2003, Ames et al. 2005, Said 2006, Dorner et al. 2007). However, explicit consideration of socially oriented, value-based objectives in setting management policy has a large impact on the degree to which public stakeholders consider those policies acceptable (Gregory 2000, Gregory and Keeney 2002, NRC 2005). This study builds on previous BN applications by integrating methods for the evaluation of science-based and value-based trade-offs into the development of water quality management portfolios.

Conclusions

In the case of Lake Champlain, this set of analyses suggests that the science-based relationships between cost, fairness, and compliance resulting from management strategies are complex, and that choosing one scenario over another will require significant value-based tradeoffs. Identifying a management scenario that controls costs, is fair, and is capable of leading to water quality restoration will be difficult in any of the study watersheds. Additionally, our results show that in some watersheds, achieving compliance with loading targets is unlikely under any management scenario, meaning it is unlikely that water quality conditions in some parts of the lake can be improved through watershed management. However, in other watersheds, high rates of compliance can be achieved, helping to ensure that water quality remains high in sections of the lake fed by those tributaries. Making good decisions about where and how to invest in restoration strategies for Lake Champlain requires careful consideration of where managers have the greatest control, and where that control can be leveraged for the greatest benefit.

The complexity involved in developing large-scale restoration plans calls for the use flexible and rigorous analytic methods that can distinguish between and address both science-based and value-based questions. We demonstrated an approach that provides managers with a foundation for good decision-making that explicitly considers uncertainty and the trade-offs among competing, value-based objectives. In turn, this approach can help managers to develop more transparent, more defensible, and more widely accepted – that is, better – restoration plans.

References

Ames, D. P., B. T. Neilson, D. K. Stevens, and U. Lall. 2005. Using Bayesian networks to model watershed management decisions: an East Canyon Creek case study. Journal of Hydroinformatics **7**:267-282.

Analytica. 2012. Analytica User Guide, version 4.5.1. Lumina Decision Systems, Inc., Los Gatos, CA.

Barton, D. N., T. Saloranta, J. Moe, H. Olav-Eggestad, N. Vagstad, A. L. Solheim, and J. R. Selvik. 2006. Using Bayesian belief networks in pollution abatement planning: examples for Morse catchment, southeastern Norway. Norwegian Institute for Water Research, Report No. SNO 5213-2006.

Borsuk, M. E., S. Schweizer, and P. Reichert. 2012. A Bayesian network model for integrative river rehabilitation planning and management. Integrated Environmental Assessment and Management **8**:462-472.

Borsuk, M. E., C. A. Stow, and K. H. Reckhow. 2003. Integrated approach to total maximum daily load development for Neuse River Estuary using Bayesian probability network model (Neu-BERN). Journal of Water Resources Planning and Management-Asce **129**:271-282.

Borsuk, M. E., C. A. Stow, and K. H. Reckhow. 2004. A Bayesian network of eutrophication models for synthesis, prediction, and uncertainty analysis. Ecological Modelling **173**:219-239.

Cerucci, M., and J. M. Conrad. 2003. The use of binary optimization and hydrologic models to form riparian buffers. Journal of the American Water Resources Association **39**:1167-1180.

Dorner, S., J. Shi, and D. Swayne. 2007. Multi-objective modelling and decision support using a Bayesian network approximation to a non-point source pollution model. Environmental Modelling & Software **22**:211-222.

EPA. 1999. Draft Guidance for Water Quality-based Decisions: The TMDL Process (Second Edition). EPA 841-D-99-001, US EPA Office of Water, Washington DC.

Farmani, R., H. J. Henriksen, D. Savic, and D. Butler. 2012. An evolutionary Bayesian belief network methodology for participatory decision making under uncertainty: An application to groundwater management. Integrated Environmental Assessment and Management **8**:456-461.

Fry, J. A., G. Xian, S. Jin, J. Dewitz, C. G. Homer, L. Yang, C. A. Barnes, N. D. Herold, and J. D. Wickham. 2011. Completion of the 2006 National Land Cover Database for the conterminous United States. Photogrammetric Engineering & Remote Sensing **77**.

Gitau, M. W., T. L. Veith, W. J. Gburek, and A. R. Jarrett. 2006. Watershed level best management practice selection and placement in the town brook watershed, New York. Journal of the American Water Resources Association **42**:1565-1581.

Goodwin, P., and G. Wright. 2004. Decision Analysis for Management Judgment: Fourth Edition. John Wiley & Sons, Ltd., West Sussex, England.

Gregory, R. 2000. Using stakeholder values to make smarter environmental decisions. Environment **42**:34-44.

Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. Structured Decision Making: A Practical Guide to Environmental Management Choices. Wiley-Blackwell.

Gregory, R., L. Failing, D. Ohlson, and T. L. McDaniels. 2006. Some pitfalls of an overemphasis on science in environmental risk management decisions. Journal of Risk Research **9**:717-735.

Gregory, R., and R. L. Keeney. 1994. Creating policy alternatives using stakeholder values. Management Science **40**:1035-1048.

Gregory, R. S., and R. L. Keeney. 2002. Making smarter environmental management decisions. Journal of the American Water Resources Association **38**:1601-1612.

Gudimov, A., E. O'Connor, M. Dittrich, H. Jarjanazi, M. E. Palmer, E. Stainsby, J. G. Winter, J. D. Young, and G. B. Arhonditsis. 2012. Continuous Bayesian network for studying the causal links between phosphorus loading and plankton patterns in Lake Simcoe, Ontario, Canada. Environmental Science & Technology **46**:7283-7292.

Hammond, J. S., R. L. Keeney, and H. Raiffa. 1999. Smart Choices: a practical guide to making better life decisions. Broadway Books, New York.

Hirsch, R. M., P. A. Hamilton, and T. L. Miller. 2006. The increasing need for science to inform policy - More than 30 years after passage of the Clean Water Act. Water Environment Research **78**:899-900.

Jensen, F. V., and T. D. Nielsen. 2007. Bayesian Networks and Decision Graphs. Springer, Berlin.

Keeney, R. L. 1992. Value-focused thinking: a path to creative decision making. Harvard University Press, Boston, MA.

Kirkwood, C. W. 1992. An overview of methods for applied decision-analysis. Interfaces **22**:28-39.

Martin, J., P. L. Fackler, J. D. Nichols, M. C. Runge, C. L. McIntyre, B. L. Lubow, M. C. McCluskie, and J. A. Schmutz. 2011. An Adaptive-Management Framework for Optimal Control of Hiking Near Golden Eagle Nests in Denali National Park. Conservation Biology **25**:316-323.

Muleta, M. K., and J. W. Nicklow. 2005. Decision support for watershed management using evolutionary algorithms. Journal of Water Resources Planning and Management-Asce **131**:35-44.

NRC. 2001. Assessing the TMDL Approach to Water Quality Management. Water Science and Technology Board, National Academies Press, Washington, D.C.

NRC. 2005. Decision Making for the Environment: Social and Behavioral Science Research Priorities. National Academies Press, Washington DC.

Radcliffe, D. E., J. Freer, and O. Schoumans. 2009. Diffuse Phosphorus Models in the United States and Europe: Their Usages, Scales, and Uncertainties. Journal of Environmental Quality **38**:1956-1967.

Reckhow, K. H., M. N. Beaulac, and J. T. Simpson. 1980. Modeling phosphorus loading and lake response under uncertainty: A manual and compilation of export coefficients. Washington D.C.

Rodriguez, H. G., J. Popp, C. Maringanti, and I. Chaubey. 2011. Selection and placement of best management practices used to reduce water quality degradation in Lincoln Lake watershed. Water Resources Research **47**.

Said, A. 2006. The implementation of a Bayesian network for watershed management decisions. Water Resources Management **20**:591-605.

Shirmohammadi, A., I. Chaubey, R. D. Harmel, D. D. Bosch, R. Munoz-Carpena, C. Dharmasri, A. Sexton, M. Arabi, M. L. Wolfe, J. Frankenberger, C. Graff, and T. M.

Sohrabi. 2006. Uncertainty in TMDL models. Transactions of the Asabe **49**:1033-1049.

Smeltzer, E. 1997. A Phosphorus Budget, Model, and Load Reduction Strategy for Lake Champlain. Submitted to US Environmental Protection Agency Region I and Region II.

State of Vermont. 2014. Vermont Lake Champlain Phosphorus TMDL Phase 1 Implementation Plan. Prepared for US EPA Region 1 by the Vermont Agency of Natural Resources, Vermont Agency of Agriculture, Food and Markets, & Vermont Agency of Transportation. Montpelier, VT, available at http://www.watershedmanagement.vt.gov/erp/champlain/. Accessed 2/18/14.

Stehman, S. V. 2013. Estimating area from an accuracy assessment error matrix. Remote Sensing of Environment **132**:202-211.

Tetra Tech. 2013. Lake Champlain BATHTUB Model Calibration Report. Prepared for U.S. EPA Region 1 - New England. Boston MA. Available from http://www.epa.gov/region1/eco/tmdl/lakechamplain.html.

Varis, O., and S. Kuikka. 1999. Learning Bayesian decision analysis by doing: lessons from environmental and natural resources management. Ecological Modelling **119**:177-195.

VT DEC. 2014. Submitted Public Comments on the Vermont Lake Champlain Phosphorus TMDL Phase 1 Implementation Plan., Availale at: http://www.watershedmanagement.vt.gov/erp/champlain/cmnts-tmdlphase1.htm. Last accessed Feb 22, 2015.

Wickham, J. D., S. V. Stehman, L. Gass, J. Dewitz, J. A. Fry, and T. G. Wade. 2013. Accuracy assessment of NLCD 2006 land cover and impervious surface. Remote Sensing of Environment **130**:294-304.

Williams, B. K. 1996. Adaptive optimization of renewable natural resources: Solution algorithms and a computer program. Ecological Modelling **93**:101-111.

CHAPTER 5. TOWARD A BETTER FUTURE FOR WATERSHED MANAGEMENT IN LAKE CHAMPLAIN

Introduction and Context

The body of work that I presented here was motivated initially by an acknowledgement on the part of the Lake Champlain Basin Program (LCBP) that despite what seems like a large amount of effort, money, and attention devoted to the problem, non-point source phosphorus loads in most tributaries – and more importantly, the regularity and severity of algae blooms in Lake Champlain – do not seem to be declining. What's perhaps more troubling is that after roughly 30 years of management and research, the management community seems to have learned, with perhaps one key exception, relatively little about the effectiveness of their basic approach.

The current approach to improving water quality in Lake Champlain (i.e., by managing land use) is centered on voluntary commitments between jurisdictions to co-manage the lake and voluntary commitments on the part of landowners to management practices that they hope will reduce phosphorus loading. While the LCBP has enjoyed good success in establishing a culture of trust and collaboration based on those voluntary commitments between New York, Vermont, and Quebec, that culture is not enough to ensure the effective management of Lake Champlain. Effective management also requires a structure for making the difficult decisions that are inherent to high-stakes problems, and a method for tracking progress to know when the problem is solved or when to change course. Most of the effort to track progress in Lake Champlain before 2008 was aimed at tracking levels of implementation in raw terms – how many acres of this, how many dollars went there – and not in terms of how much work needed to be done or whether the work was having its intended effect. Unfortunately, what the public ultimately cares about is not how many rain barrels have been installed, but instead whether they can enjoy time spent on Missisquoi Bay in August. Despite all of those implementation efforts, the answer is still "No", and the tracking data that exists doesn't provide any answers about why.

With my work, I intended to provide analyses and tools that could help fill three major holes in the current approach to managing Lake Champlain:

1) an ability to quantitatively track management progress, relative to what could be done, alongside ecologically and socially meaningful outcomes,
2) an analytically rigorous way of accounting for the limited understanding of the connection between watersheds and the lake and therefore limited ability to predict the effects of management actions taken, and
3) an approach for evaluating technical trade-offs alongside value-based trade-offs in a logically consistent way.

The Phosphorus Indicator table presented in Chapter 2 fills the first of the holes to an acceptable level. It provides a structure for a clear evaluation of the amount of work that has occurred relative to the realm of possibility and relative to other sectors. That context is important for communicating to the rest of the management community and to the public the real scale of effort that has been expended. Without an understanding of how much could be done, it's impossible to know whether the States of New York and Vermont are really doing all that they can for water quality improvements. Though valuable in many ways, the Phosphorus Indicator approach falls short of including outcomes that are truly socially meaningful, and also fails to provide a clear and quantitative connection between management implementation and ecological response.

Making defensible decisions requires honesty and humility about the decision outcomes, and the techniques presented in Chapters 3 and 4 provide that ability. Though the outcome of this work is a more honest appraisal of the likely effects of management approaches, the take-home message is that our degree of control over phosphorus loads is likely to be much smaller than we've believed and is not promising for the future of the Lake. Taken together, Chapters 3 and 4 address the shortcomings of the Phosphorus Indicator approach, and borrowing insights and methods from behavioral psychology and decision analysis, draw out the connection between management actions and their ecologically- and socially-relevant outcomes while incorporating the uncertainty in those connections. While I did provide a framework for incorporating other values into the decision-process, the applicability of that framework does assume that the State of Vermont knows what its organizational values are, and that transparency about those values is positive. Additional work on the part of the State to explore its values related to the restoration of water quality in Lake Champlain would be worthwhile as they begin to develop implementation plans for the new TMDL.

Collectively, this work provides a coherent structure for making defensible decisions in light of our relatively poor understanding about the effectiveness of controlling NPS pollution, and additionally, sheds some light onto the issue of how much control the management community might actually have. Despite these important advances there are still many questions left unanswered about the connections between watershed management and lake water quality. Each of these questions individually could have profound implications for how watershed management happens around the country, and specifically for the relevance of the TMDL targets for Lake Champlain and for the utility of the decision support approaches I've developed. Answers to these questions – or at least acknowledgement of the lack of answers and accommodation of the existing uncertainty –

are critical to the success of water quality restoration efforts in Lake Champlain. I briefly lay out three of these questions below as areas of worthwhile future investigation.

Future Directions

1. What is the connection between watershed phosphorus loads, in-lake concentrations, and algae blooms?

Evidence accumulated over the past decade has cast doubt on the assumption that watershed loading is the most important contributor to in-lake phosphorus concentrations under eutrophic and mesotrophic conditions (Smith et al. 2011, Sondergaard et al. 2013). Additionally, though the phosphorus limitation paradigm forms much of the basis for reducing phosphorus loads as a method to control algae blooms (Schindler et al. 2008), evidence for its widespread validity is equivocal (Sterner 2008). In fact, local evidence suggests that ratios of nitrogen to phosphorus and in-lake sediment anoxia may be more important for explaining algae bloom patterns for parts of Lake Champlain (Pearce et al. 2013), lending support to the possibility that phosphorus reduction alone will not control algae blooms. Reversing eutrophication in similarly shallow and eutrophic lakes in Scandinavia has required food web manipulations in addition to reducing tributary phosphorus loads to overcome the chemical and biological resilience that maintains eutrophic conditions (Jeppesen et al. 2012).

Intuitively, it seems obvious that reducing tributary loads is the key to reducing in-lake phosphorus concentrations and algae blooms in the long term. However, a lack of understanding about the short-term connections between these three factors obscures understanding about whether watershed management can be effective for reversing the effects of eutrophication.

2. What is the role of natural hydrologic variability in determining phosphorus loads, and how should we account for shifting sources and scales of variability?

This question has two elements that are important for managing watershed-scale nutrient loads. The first is that most of the management practices that attempt to slow water movement across the landscape are based upon assumptions about the amount and pattern of precipitation across that landscape. However, it is becoming more and more clear that these regimes are changing in Lake Champlain (Figure 5-1), and around the world, in part because of climate changes (Milly et al. 2008), but also because of land use changes (Hirsch 2011). Therefore, the non-stationary effects of runoff generation, and the connections between runoff generation and nutrient loads are becoming more and more difficult to predict using the standard analytical tools, which in turn makes learning about the effects of watershed management very difficult.

Part of the difficulty in measuring the effects of watershed management within the watersheds themselves is the effect of the time lags between the implementation of practices and observing the effects downstream. These time lags are driven by highly variable sediment transport (Schmelter and Stevens 2013) and nutrient spiraling processes (Ensign and Doyle 2006), and affect both the particulate and dissolved phosphorus portions in streams. Though predictability of these processes is understandably poor, time lags are generally considered to be very long for watershed-scale nutrient and sediment loads and have been identified as a likely culprit in delaying the attainment of phosphorus reductions at the watershed scale in the Lake Champlain Basin (Meals et al. 2010).

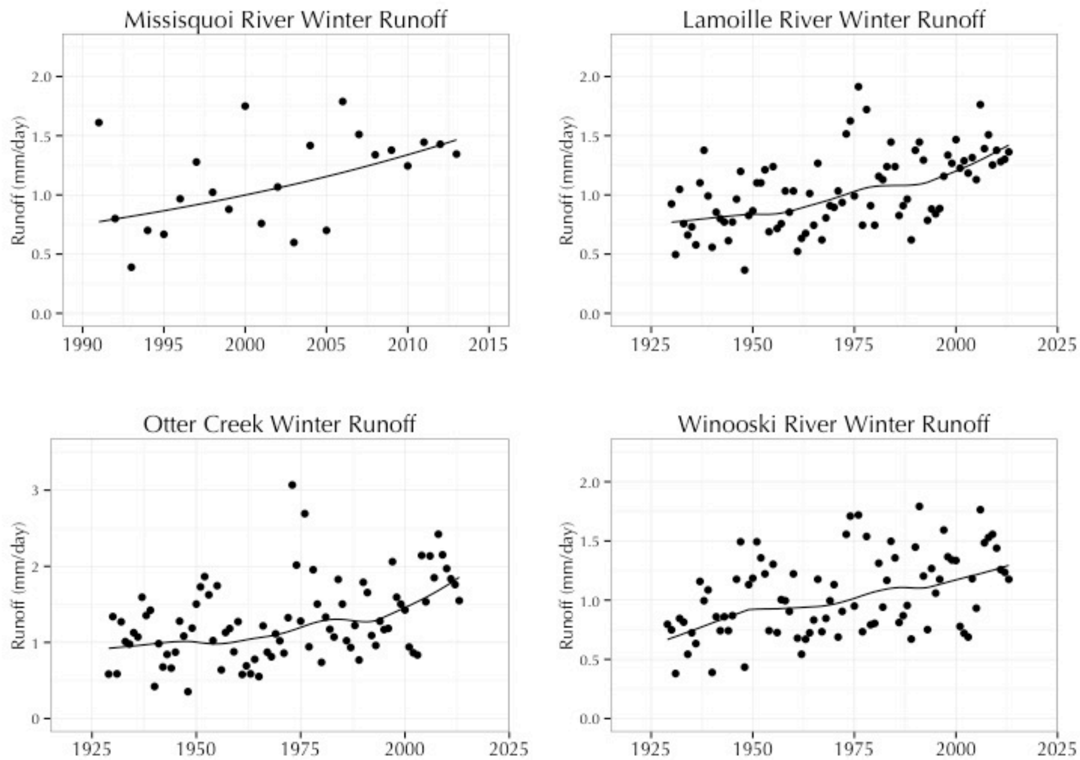

Figure 5-1. Watershed runoff for four watersheds in the Lake Champlain basin showing substantial increase in median daily runoff for the winter months December through March inclusive. Note the shorter discharge record for the Missisquoi River. Data are from USGS gauges, processed using EGRET (Hirsch and De Cicco 2014).

3. Is the structure of the TMDL program conducive to effective NPS phosphorus management?

The National Research Council's review of the scientific basis of the TMDL program identified several formidable sources of uncertainty in the way that water quality standards and allocations to achieve those standards were set (NRC 2001). The report also made many suggestions for changes to the structure of the TMDL program that would address and decrease those sources of uncertainty over time. While some of the recommendations have been enacted, many of the most important ones from a decision making perspective remain unaddressed and, contrary to the intention of the program, may actually hinder effective management of water quality. In particular, the continued use of proxy management targets, continued reliance on ecological mechanisms over which managers have little control, and continued use of complex mechanistic models that do not explicitly account for the uncertainty in important processes and relationships are all problematic. These elements of the TMDL program will continue to reduce the usefulness of decision support approaches such as I've described in this dissertation because without direct cause and effect linkages and clearly articulated and meaningful goals, good decision making is impossible.

The field of decision analysis is clear about the benefits of a process that focuses on outcomes that are actually important, that evaluates alternatives that are within the sphere of influence, and that incorporates uncertainty into decisions where the risk of bad outcomes is serious (von Winterfeldt and Edwards 1986). Analytically focused decision support tools are built under the assumption that the specified management objectives are correct and are actually relevant to the underlying problem. If those assumptions are not correct, the utility of the tools is minimal.

One final well-cited recommendation from the NRC report was that the TMDL program should embrace and encourage a style of implementation based on the concepts of adaptive management. Relatively recent work has identified adaptive elements of some TMDL implementations, but a full process has yet to be developed and tested (Freedman et al. 2008). Given the decision support approach outlined here and a fairly robust monitoring program, the Lake Champlain Basin could become an excellent case for testing the adaptive implementation process and for learning how to push watershed management in a new and positive direction.

References

Ensign, S. H., and M. W. Doyle. 2006. Nutrient spiraling in streams and river networks. Journal of Geophysical Research-Biogeosciences **111**.

Freedman, P., K. Reckhow, L. Shabman, J. Benaman, R. Schwer, and T. Stiles. 2008. A new approach to adaptive implementation in the TMDL program. Water Practice **2**.

Hirsch, R. M. 2011. A Perspective on Nonstationarity and Water Management. JAWRA Journal of the American Water Resources Association **47**:436-446.

Hirsch, R. M., and L. A. De Cicco. 2014. User guide to Exploration and Graphics for RivEr Trends (EGRET) and dataRetrieval - R packages for hydrologic data (version 2.0, February, 2015).

Jeppesen, E., M. Sondergaard, T. L. Lauridsen, T. A. Davidson, Z. W. Liu, N. Mazzeo, C. Trochine, K. Ozkan, H. S. Jensen, D. Trolle, F. Starling, X. Lazzaro, L. S. Johansson, R. Bjerring, L. Liboriussen, S. E. Larsen, F. Landkildehus, S. Egemose, and M. Meerhoff. 2012. Biomanipulation as a Restoration Tool to Combat Eutrophication: Recent Advances and Future Challenges. Pages 411-488 *in* G. Woodward, U. Jacob, and E. J. Ogorman, editors. Advances in Ecological Research, Vol 47: Global Change in Multispecies Systems, Pt 2. Elsevier Academic Press Inc, San Diego.

Meals, D. W., S. A. Dressing, and T. E. Davenport. 2010. Lag Time in Water Quality Response to Best Management Practices: A Review. Journal of Environmental Quality **39**:85-96.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change - Stationarity is dead: Whither water management? Science **319**:573-574.

NRC. 2001. Assessing the TMDL Approach to Water Quality Management. Water Science and Technology Board, National Academies Press, Washington, D.C.

Pearce, A. R., D. M. Rizzo, M. C. Watzin, and G. K. Druschel. 2013. Unraveling Associations between Cyanobacteria Blooms and In-Lake Environmental Conditions in Missisquoi Bay, Lake Champlain, USA, Using a Modified Self-Organizing Map. Environmental Science & Technology **47**:14267-14274.

Schindler, D. W., R. E. Hecky, D. L. Findlay, M. P. Stainton, B. R. Parker, M. J. Paterson, K. G. Beaty, M. Lyng, and S. E. M. Kasian. 2008. Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. Proceedings of the National Academy of Sciences **105**:11254-11258.

Schmelter, M. L., and D. K. Stevens. 2013. Traditional and Bayesian Statistical Models in Fluvial Sediment Transport. Journal of Hydraulic Engineering-Asce **139**:336-340.

Smith, L., M. C. Watzin, and G. Druschel. 2011. Relating sediment phosphorus mobility to seasonal and diel redox fluctuations at the sediment-water interface in a eutrophic freshwater lake. Limnology and Oceanography **56**:2251-2264.

Sondergaard, M., R. Bjerring, and E. Jeppesen. 2013. Persistent internal phosphorus loading during summer in shallow eutrophic lakes. Hydrobiologia **710**:95-107.

Sterner, R. W. 2008. On the Phosphorus Limitation Paradigm for Lakes. International Review of Hydrobiology **93**:433-445.

von Winterfeldt, D., and W. Edwards. 1986. Decision Analysis and Behavioral Research. Cambridge University Press, London.

COMPREHENSIVE BIBLIOGRAPHY

Allan, C., and A. Curtis. 2005. Nipped in the bud: Why regional scale adaptive management is not blooming. Environmental Management **36**:414-425.

Allan, J. D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology Evolution and Systematics **35**:257-284.

Allen, C. R., J. J. Fontaine, K. L. Pope, and A. S. Garmestani. 2011. Adaptive management for a turbulent future. Journal of Environmental Management **92**:1339-1345.

Allen, C. R., and L. H. Gunderson. 2011. Pathology and failure in the design and implementation of adaptive management. Journal of Environmental Management **92**:1379-1384.

Ames, D. P., B. T. Neilson, D. K. Stevens, and U. Lall. 2005. Using Bayesian networks to model watershed management decisions: an East Canyon Creek case study. Journal of Hydroinformatics **7**:267-282.

Analytica. 2012. Analytica User Guide, version 4.5.1. Lumina Decision Systems, Inc., Los Gatos, CA.

Anderson, B. H., and F. R. Magdoff. 2000. Dairy farm characteristics and managed flows of phosphorus. American Journal of Alternative Agriculture **15**:19-25.

Arabi, M., D. W. Meals, and D. L. Hoag. 2012. Watershed Modeling: National Institute of Food and Agriculture - Conservation Effects Assessment Project.*in* D. L. Osmond, D. W. Meals, D. L. Hoag, and M. Arabi, editors. How to Build Better Agricultural Conservation Programs to Protect Water Quality: The National Institute of Food and Agriculture – Conservation Effects Assessment Project Experience. Soil and Water Conservation Society, Ankeny, IA.

Barth, C. A. 1995. Nutrient Movement from the Lawn to the Stream. Pages 239-249 Watershed Protection Techniques. Center for Watershed Protection.

Barton, D. N., T. Saloranta, J. Moe, H. Olav-Eggestad, N. Vagstad, A. L. Solheim, and J. R. Selvik. 2006. Using Bayesian belief networks in pollution abatement planning: examples for Morse catchment, southeastern Norway. Norwegian Institute for Water Research, Report No. SNO 5213-2006.

Blumstein, D. T. 2007. Darwinian decision making: Putting the adaptive into adaptive management. Conservation Biology **21**:552-553.

Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems: Degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association **33**:1077-1090.

Borah, D. K., G. Yagow, A. Saleh, P. L. Barnes, W. Rosenthal, E. C. Krug, and L. M. Hauck. 2006. Sediment and nutrient modeling for TMDL development and implementation. Transactions of the Asabe **49**:967-986.

Borisova, T., J. Shortle, R. D. Horan, and D. Abler. 2005. Value of information for water quality management. Water Resources Research **41**.

Borsuk, M. E., S. Schweizer, and P. Reichert. 2012. A Bayesian network model for integrative river rehabilitation planning and management. Integrated Environmental Assessment and Management **8**:462-472.

Borsuk, M. E., C. A. Stow, and K. H. Reckhow. 2003. Integrated approach to total maximum daily load development for Neuse River Estuary using Bayesian

probability network model (Neu-BERN). Journal of Water Resources Planning and Management-Asce **129**:271-282.
Borsuk, M. E., C. A. Stow, and K. H. Reckhow. 2004. A Bayesian network of eutrophication models for synthesis, prediction, and uncertainty analysis. Ecological Modelling **173**:219-239.
Broad, S. T., and R. Corkrey. 2011. Estimating annual generation rates of total P and total N for different land uses in Tasmania, Australia. Journal of Environmental Management **92**:1609-1617.
Budd, L. F., and D. W. Meals. 1994. Lake Champlain Nonpoint Source Pollution Assessment. Technical Report No. 6A. Lake Champlain Management Conference, Grand Isle, VT.
Bushey, J. K., J. E. Carter, J. R. Chamberlain, and S. A. Flis. 2009. Reducting Phosphorus Runoff from Small Livestock Farms into Missisquoi Bay. Technical Report No. 58. Lake Champlain Basin Program, Grand Isle, VT.
Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen. Ecological Applications **8**:559-568.
Carriger, J. F., W. S. Fisher, T. B. Stockton, and P. E. Sturm. 2013. Advancing the Guánica Bay (Puerto Rico) Watershed Management Plan. Coastal Management **41**:19-38.
Caulkins, J. P. 2002. Using models that incorporate uncertainty. Journal of Policy Analysis and Management **21**:486-491.
Cerosaletti, P. E., D. G. Fox, and L. E. Chase. 2004. Phosphorus reduction through precision feeding of dairy cattle. Journal of Dairy Science **87**:2314-2323.
Cerucci, M., and J. M. Conrad. 2003. The use of binary optimization and hydrologic models to form riparian buffers. Journal of the American Water Resources Association **39**:1167-1180.
Cha, Y., C. A. Stow, K. H. Reckhow, C. DeMarchi, and T. H. Johengen. 2010. Phosphorus load estimation in the Saginaw River, MI using a Bayesian hierarchical/multilevel model. Water Research **44**:3270-3282.
Contach, K., C. Ballard, W. Emerich, C. Sniffen, and E. Thomas. 2003. The Feeding of Supplemental Phosphorus on Dairy Farms in the Lake Champlain Basin: An Education/Demonstration Project. Technical Report No. 42. Lake Champlain Basin Program, Grand Isle, VT.
Darby, H., P. Halteman, and D. Heleba. 2013. The Effectiveness of Nutrient Management Plans on Vermont Dairy Farms. Journal of Extension [On-line] **(in press)**.
Davie, D. K., and C. L. Lant. 1994. The effect of CRP enrollement on sediment loads in two southern Illinois streams. Journal of Soil and Water Conservation **49**:407-412.
Dorner, S., J. Shi, and D. Swayne. 2007. Multi-objective modelling and decision support using a Bayesian network approximation to a non-point source pollution model. Environmental Modelling & Software **22**:211-222.
Dou, Z. X., K. F. Knowlton, R. A. Kohn, Z. G. Wu, L. D. Satter, G. Y. Zhang, J. D. Toth, and J. D. Ferguson. 2002. Phosphorus characteristics of dairy feces affected by diets. Journal of Environmental Quality **31**:2058-2065.
Eberhard, R., C. J. Robinson, J. Waterhouse, J. Parslow, B. Hart, R. Grayson, and B. Taylor. 2009. Adaptive management for water quality planning - from theory to practice. Marine and Freshwater Research **60**:1189-1195.

Ensign, S. H., and M. W. Doyle. 2006. Nutrient spiraling in streams and river networks. Journal of Geophysical Research-Biogeosciences **111**.

EPA. 1999a. Combined Sewer Overflow Management Fact Sheet Sewer Separation. EPA 832-F-99-041, US EPA Office of Water, Washington DC.

EPA. 1999b. Draft Guidance for Water Quality-based Decisions: The TMDL Process (Second Edition). EPA 841-D-99-001, US EPA Office of Water, Washington DC.

EPA. 2000. Evaluation Guidelines for Ecological Indicators. US EPA Office of Research and Development, Washington DC.

ESRI. 2012. ArcGIS 10.1. Environmental Systems Research Institute, Redlands, CA.

Farmani, R., H. J. Henriksen, D. Savic, and D. Butler. 2012. An evolutionary Bayesian belief network methodology for participatory decision making under uncertainty: An application to groundwater management. Integrated Environmental Assessment and Management **8**:456-461.

Field, R., and E. J. Struzeski, Jr. 1972. Management and Control of Combined Sewer Overflows. Journal (Water Pollution Control Federation) **44**:1393-1415.

Ford, R. F. 2012. Agricultural Phosphorus in Vermont's Missisquoi Bay Watershed: History, Status, and Solutions. University of Vermont, Burlington, VT.

Freedman, P., K. Reckhow, L. Shabman, J. Benaman, R. Schwer, and T. Stiles. 2008. A new approach to adaptive implementation in the TMDL program. Water Practice **2**.

Fry, J. A., G. Xian, S. Jin, J. Dewitz, C. G. Homer, L. Yang, C. A. Barnes, N. D. Herold, and J. D. Wickham. 2011. Completion of the 2006 National Land Cover Database for the conterminous United States. Photogrammetric Engineering & Remote Sensing **77**.

Gaddis, E. J. B., and A. Voinov. 2010. Spatially Explicit Modeling of Land Use Specific Phosphorus Transport Pathways to Improve TMDL Load Estimates and Implementation Planning. Water Resources Management **24**:1621-1644.

Garton, J. 2013. Personal communication, Nov. 12, 2013.

Gburek, W. J., E. Barberis, P. M. Haygarth, B. Kronvang, and C. Stamm. 2005. Phosphorus Mobility in the Landscape *in* J. T. Sims and A. N. Sharpley, editors. Phosphorus: Agriculture and the Environment. American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Pubs., Madison, WI.

Gelman, A., and J. Hill. 2009. Data Analysis Using Regression and Multilevel/Hierarchical Models. Cambridge University Press, New York, NY.

Geohring, L. D., T. S. Steenhuis, A. S. Brooks, M. N. Rosenwald, J. Chen, and V. J. Putman. 1999. Cost-Effective Phosphorus Removal from Secondary Wastewater Effluent through Mineral Adsorption. Technical Report No. 36. Lake Champlain Basin Program, Grand Isle, VT.

Ghebremichael, L. T., T. L. Veith, and M. C. Watzin. 2010. Determination of Critical Source Areas for phosphorus loss: Lake Champlain Basin, Vermont. Transactions of the Asabe **53**:1595-1604.

Ghebremichael, L. T., and M. C. Watzin. 2010. An Environmental Accounting System to Track Nonpoint Source Phosphorus Pollution in the Lake Champlain Basin, Technical Report No. 60. Lake Champlain Basin Program, Grand Isle, VT.

Gigerenzer, G. 2014. Risk Savvy: How to make good decisions. Penguin.

Gitau, M. W., T. L. Veith, W. J. Gburek, and A. R. Jarrett. 2006. Watershed level best management practice selection and placement in the town brook watershed, New York. Journal of the American Water Resources Association **42**:1565-1581.

Goodwin, P., and G. Wright. 2004. Decision Analysis for Management Judgment: Fourth Edition. John Wiley & Sons, Ltd., West Sussex, England.

Gregory, R. 2000. Using stakeholder values to make smarter environmental decisions. Environment **42**:34-44.

Gregory, R. 2013. Decision-Aiding for the Lake Champlain Basin Program: Report of a Two-Day Program Overview, February 26/27, 2013. Prepared for the Lake Champlain Basin Program.

Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. Structured Decision Making: A Practical Guide to Environmental Management Choices. Wiley-Blackwell.

Gregory, R., L. Failing, D. Ohlson, and T. L. McDaniels. 2006a. Some pitfalls of an overemphasis on science in environmental risk management decisions. Journal of Risk Research **9**:717-735.

Gregory, R., and R. L. Keeney. 1994. Creating policy alternatives using stakeholder values. Management Science **40**:1035-1048.

Gregory, R., D. Ohlson, and J. Arvai. 2006b. Deconstructing adaptive management: Criteria for applications to environmental management. Ecological Applications **16**:2411-2425.

Gregory, R. S., and R. L. Keeney. 2002. Making smarter environmental management decisions. Journal of the American Water Resources Association **38**:1601-1612.

Gregory, S., A. W. Allen, M. Baker, K. Boyer, T. Dillaha, and J. Elliot. 2007. Realistic expectations of timing between conservation and restoration actions and ecological responses *in* M. Schnepf and C. Cox, editors. Managing Agricultural Landscapes for Environmental Quality: Strengthening the Science Base. Soil and Water Conservation Society, Ankeny, IA.

Gudimov, A., E. O'Connor, M. Dittrich, H. Jarjanazi, M. E. Palmer, E. Stainsby, J. G. Winter, J. D. Young, and G. B. Arhonditsis. 2012. Continuous Bayesian network for studying the causal links between phosphorus loading and plankton patterns in Lake Simcoe, Ontario, Canada. Environmental Science & Technology **46**:7283-7292.

Gunderson, L., C. S. Holling, and S. S. Light, editors. 1995. Barriers and Bridges to the Renewal of Ecosystems and Institutions. Columbia University Press, New York.

Gunderson, L. H. 2001. Managing surprising ecosystems in southern Florida. Ecological Economics **37**:371-378.

Hammond, J. S., R. L. Keeney, and H. Raiffa. 1998. Even Swaps: A Rational Method for Making Trade-offs. Harvard Business Review.

Hammond, J. S., R. L. Keeney, and H. Raiffa. 1999. Smart Choices: a practical guide to making better life decisions. Broadway Books, New York.

Heath, G. R., G. M. Grant, and R. S. Seymour. 2004. Combined sewer separation not necessarily the answer to CSO control. Proceedings of the Water Environment Federation **2004**:414-435.

Hegman, W., D. Wang, and C. Borer. 1999. Estimation of Lake Champlain Basinwide Nonpoint Source Phosphorus Export. Technical Report No.31. Lake Champlain Basin Program, Grand Isle, VT.

Hirsch, R. M. 2011. A Perspective on Nonstationarity and Water Management. JAWRA Journal of the American Water Resources Association **47**:436-446.

Hirsch, R. M., and L. A. De Cicco. 2014. User guide to Exploration and Graphics for RivEr Trends (EGRET) and dataRetrieval - R packages for hydrologic data (version 2.0, February, 2015).
Hirsch, R. M., P. A. Hamilton, and T. L. Miller. 2006. The increasing need for science to inform policy - More than 30 years after passage of the Clean Water Act. Water Environment Research **78**:899-900.
Hirsch, R. M., D. L. Moyer, and S. A. Archfield. 2010. Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay river inputs. JAWRA Journal of the American Water Resources Association **46**:857-880.
Hoffmann, J. P., E. A. Cassell, J. C. Drake, S. N. Levine, D. W. Meals, and D. Wang. 1996. Understanding Phosphorus Cycling, Transport and Storage in Stream Ecosystems as a Basis for Phosphorus Management. Technical Report No. 20. Lake Champlain Basin Program, Grand Isle, VT.
Holling, C. S., editor. 1978. Adaptive Environmental Assessment and Management. John Wiley & Sons, Chichester, New York, Brisbane, Toronto.
Hughes, J. W., W. E. Jokela, D. Wang, and C. Borer. 1999. Determination and Quantification of Factors Controlling Pollutant Delivery from Agricultural Land to Streams in the Lake Champlain Basin. Technical Report No. 35. Lake Champlain Basin Program, Grand Isle, VT.
Jensen, F. V., and T. D. Nielsen. 2007. Bayesian Networks and Decision Graphs. Springer, Berlin.
Jeppesen, E., M. Sondergaard, T. L. Lauridsen, T. A. Davidson, Z. W. Liu, N. Mazzeo, C. Trochine, K. Ozkan, H. S. Jensen, D. Trolle, F. Starling, X. Lazzaro, L. S. Johansson, R. Bjerring, L. Liboriussen, S. E. Larsen, F. Landkildehus, S. Egemose, and M. Meerhoff. 2012. Biomanipulation as a Restoration Tool to Combat Eutrophication: Recent Advances and Future Challenges. Pages 411-488 *in* G. Woodward, U. Jacob, and E. J. Ogorman, editors. Advances in Ecological Research, Vol 47: Global Change in Multispecies Systems, Pt 2. Elsevier Academic Press Inc, San Diego.
Jin, S. M., L. M. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment **132**:159-175.
Johnes, P. J. 1996. Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. Journal of Hydrology **183**:323-349.
Johnson, C. 1998. The Nature of Vermont. University Press of New England, Lebanon, NH.
Johnson, F., and K. Williams. 1999. Protocol and practice in the adaptive management of waterfowl harvests. Conservation Ecology **3**:article 8. [online] URL: http://www.ecologyandsociety.org/vol3/iss1/art8/.
Johnson, F. A., W. L. Kendall, and J. A. Dubovsky. 2002. Conditions and limitations on learning in the adaptive management of mallard harvests. Wildlife Society Bulletin **30**:176-185.
Jokela, W. E., J. P. Tilley, and D. S. Ross. 2010. Manure Nutrient Content on Vermont Dairy Farms: Long-Term Trends and Relationships. Communications in Soil Science and Plant Analysis **41**:623-637.
Kangas, A. S., P. Horne, and P. Leskinen. 2010. Measuring the Value of Information in Multicriteria Decisionmaking. Forest Science **56**:558-566.
Keeney, R. L. 1982. Decision-analysis - an overview. Operations Research **30**:803-838.

Keeney, R. L. 1992. Value-focused thinking: a path to creative decision making. Harvard University Press, Boston, MA.
Kirkwood, C. W. 1992a. Estimating the impact of uncertainty on a determinisitc multiattribute evaluation. Management Science **38**:819-826.
Kirkwood, C. W. 1992b. An overview of methods for applied decision-analysis. Interfaces **22**:28-39.
Klebs, P. 2008. Optimizing biological phosphorus removal from an SBR system - Middlebury, VT. Aqua-Aerobic Systems, Inc., Loves Park, IL.
Kline, M., and B. Cahoon. 2008. Restoring Ecological Processes in Protected River Corridors. River Management Program, Vermont Agency of Natural Resources, Waterbury, VT.
Lake Champlain Basin Program. 1996. Opportunities for Action: An Evolving Plan for the Future of the Lake Champlain Basin. Lake Champlain Basin Program, Grand Isle, VT.
Lake Champlain Basin Program. 2000. Preliminary Evaluation of Progress Toward Lake Champlain Basin Program Phosphorus Reduction Goals. Lake Champlain Basin Program, Grand Isle, VT.
Lake Champlain Basin Program. 2003. Opportunities for Action: An Evolving Plan for the Future of the Lake Champlain Basin. Lake Champlain Basin Program, Grand Isle, VT.
Lake Champlain Basin Program. 2010. Opportunities for Action: An Evolving Plan for the Future of the Lake Champlain Basin. Lake Champlain Basin Program, Grand Isle, VT.
Lake Champlain Basin Program. 2012. Publications Database Website. URL: http://www.lcbp.org/publication_search.aspx. Last Accessed: 01/18/2012.
Lake Champlain Study. 1979. Shaping the Future of Lake Champlain. Final Report. Burlington, VT.
Langendoen, E. J., A. Simon, L. Klimetz, N. Bankhead, and M. E. Ursic. 2012. Quantifying Sediment Loadings from Streambank Erosion in Selected Agricultural Watersheds Draining to Lake Champlain. Technical Report No. 72. Lake Champlain Basin Program, Grand Isle, VT.
LCBP. 2008. State of the Lake and Ecosystem Indicators Report. Lake Champlain Basin Program, Grand Isle, VT.
LCBP. 2012. State of the Lake and Ecosystem Indicators Report. Lake Champlain Steering Committee, Lake Champlain Basin Program, Grand Isle, VT.
LCLGRPB. 2012. Lake Champlain watershed water quality management planning: roadside erosion assessment and inventory. Lake Champlain-Lake George Regional Planning Board, Lake George, NY.
Lempert, R. J., S. W. Popper, and S. C. Bankes. 2003. Shaping the next one hundred years: New methods for quantitative, long-term policy analysis. RAND-Pardee Center, Santa Monica, CA.
Levine, S. N., A. D. Shambaugh, S. E. Pomeroy, and M. Braner. 1997. Phosphorus, nitrogen, and silica as controls on phytoplankton biomass and species composition in Lake Champlain (USA-Canada). Journal of Great Lakes Research **23**:131-148.
Liu, M., and J. Lu. 2013. Solution of export coefficients of nitrogen from different land-use patterns based on Bayesian analysis. Water Science and Technology **68**:632-640.

Maguire, R. O., G. L. Mullins, and M. Brosius. 2008. Evaluating long-term nitrogen- versus phosphorus-based nutrient management of poultry litter. Journal of Environmental Quality **37**:1810-1816.

Marcot, B. G. 1998. Selecting Appropriate Statistical Procedures and Asking the Right Questions: A Synthesis *in* V. Sit and B. Taylor, editors. Statistical Methods for Adaptive Management Studies. B.C. Ministry of Forests, Research Branch, Victoria, B.C.

Martin, J., P. L. Fackler, J. D. Nichols, M. C. Runge, C. L. McIntyre, B. L. Lubow, M. C. McCluskie, and J. A. Schmutz. 2011. An Adaptive-Management Framework for Optimal Control of Hiking Near Golden Eagle Nests in Denali National Park. Conservation Biology **25**:316-323.

McCann, R. K., B. G. Marcot, and R. Ellis. 2006. Bayesian belief networks: applications in ecology and natural resource management. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere **36**:3053-3062.

McCarthy, M. A., and H. P. Possingham. 2007. Active adaptive management for conservation. Conservation Biology **21**:956-963.

McDowell, R. W., A. N. Sharpley, and A. T. Chalmers. 2002. Land use and flow regime effects on phosphorus chemical dynamics in the fluvial sediment of the Winooski River, Vermont. Ecological Engineering **18**:477-487.

Meals, D. W. 1996. Watershed-scale response to agricultural diffuse pollution control programs in Vermont, USA. Water Science and Technology **33**:197-204.

Meals, D. W. 2001. Water quality response to riparian restoration in an agricultural watershed in Vermont, USA. Water Science and Technology **43**:175-182.

Meals, D. W., and L. F. Budd. 1998. Lake Champlain Basin nonpoint source phosphorus assessment. Journal of the American Water Resources Association **34**:251-265.

Meals, D. W., S. A. Dressing, and T. E. Davenport. 2010. Lag Time in Water Quality Response to Best Management Practices: A Review. Journal of Environmental Quality **39**:85-96.

Medalie, L. 2013. Concentration, flux, and the analysis of trends of total and dissolved phosphorus, total nitrogen, and chloride in 18 tributaries to Lake Champlain, Vermont and New York, 1990–2011., U.S. Geological Survey Scientific Investigations Report 2013–5021.

Medalie, L., R. M. Hirsch, and S. A. Archfield. 2012. Use of flow-normalization to evaluate nutrient concentration and flux changes in Lake Champlain tributaries, 1990–2009. Journal of Great Lakes Research.

Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society **24**:602-612.

Michaud, A. R., I. Beaudin, J. Deslandes, F. Bonn, and C. A. Madramootoo. 2007. SWAT-predicted influence of different landscape and cropping system alterations on phosphorus mobility within the Pike River watershed of south-western Quebec. Canadian Journal of Soil Science **87**:329-344.

Mihuc, T. B., F. Dunlap, C. Binggeli, L. Myers, C. Pershyn, A. Groves, and A. Waring. 2012. Long-term patterns in Lake Champlain's zooplankton: 1992–2010. Journal of Great Lakes Research.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change - Stationarity is dead: Whither water management? Science **319**:573-574.

Morgan, M. G., and M. Henrion. 1990. Uncertainty. Cambridge University Press, Cambridge.
Muleta, M. K., and J. W. Nicklow. 2005. Decision support for watershed management using evolutionary algorithms. Journal of Water Resources Planning and Management-Asce **131**:35-44.
National Research Council. 2001. Nutrient Requirements of Dairy Cattle: Seventh Revised Edition.*in* Subcommittee on Dairy Cattle Nutrition Committee on Animal Nutrition, editor. National Academies Press, Washington, D.C.
Nichols, J. D., M. C. Runge, F. A. Johnson, and B. K. Williams. 2007. Adaptive harvest management of North American waterfowl populations: a brief history and future prospects. Journal of Ornithology **148**:S343-S349.
Niemitz, J., C. Haynes, and G. Lasher. 2013. Legacy sediments and historic land use: Chemostratigraphic evidence for excess nutrient and heavy metal sources and remobilization. Geology **41**:47-50.
NRC. 2001. Assessing the TMDL Approach to Water Quality Management. Water Science and Technology Board, National Academies Press, Washington, D.C.
NRC. 2005. Decision Making for the Environment: Social and Behavioral Science Research Priorities. National Academies Press, Washington DC.
NRC. 2007. Models in Environmental Regulatory Decision Making. National Academies Press, Washington, D.C.
O'Neil-Dunne, J. 2013. Mapping Impervious Surfaces in the Lake Champlain Basin. Technical Report No. 76. Lake Champlain Basin Program, Grand Isle, VT.
Olofsson, P., G. M. Foody, S. V. Stehman, and C. E. Woodcock. 2013. Making better use of accuracy data in land change studies: Estimating accuracy and area and quantifying uncertainty using stratified estimation. Remote Sensing of Environment **129**:122-131.
Osmond, D. L., D. W. Meals, D. L. Hoag, M. Arabi, A. E. Luloff, G. D. Jennings, M. L. McFarland, J. Spooner, A. N. Sharpley, and D. E. Line. 2012. Synthesizing the Experience of the 13 National Institute of Food and Agriculture - Conservation Effects Assessment Project Watershed Studies: Present and Future.*in* D. L. Osmond, D. W. Meals, D. L. Hoag, and M. Arabi, editors. How to Build Better Agricultural Conservation Programs to Protect Water Quality: The National Institute of Food and Agriculture - Conservation Effects Assessment Project Experience. Soil and Water Conservation Society, Ankeny, IA.
Owens, P. N. 2009. Adaptive management frameworks for natural resource management at the landscape scale: implications and applications for sediment resources. Journal of Soils and Sediments **9**:578-593.
Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics **32**:333-365.
Pearce, A. R., D. M. Rizzo, M. C. Watzin, and G. K. Druschel. 2013. Unraveling Associations between Cyanobacteria Blooms and In-Lake Environmental Conditions in Missisquoi Bay, Lake Champlain, USA, Using a Modified Self-Organizing Map. Environmental Science & Technology **47**:14267-14274.
Perkins, E. 2013. Personal communication, Nov. 6, 2013.
Peterson, G. D., G. S. Cumming, and S. R. Carpenter. 2003. Scenario planning: a tool for conservation in an uncertain world. Conservation Biology **17**:358-366.
Phillips, P., and A. Chalmers. 2009. Wastewater effluent, combined sewer overflows, and other sources of organic compounds to Lake Champlain. Journal of the American Water Resources Association **45**:45-57.

Pilkey-Jarvis, L., and O. H. Pilkey. 2008. Useless Arithmetic: Ten Points to Ponder When Using Mathematical Models in Environmental Decision Making. Public Administration Review **68**:470-479.

Plummer, M. 2013. JAGS version 3.4.0 manual.

Plummer, M. 2014. rjags: Bayesian graphical models using MCMC. R package version 3-14. http://CRAN.R-project.org/package=rjags.

Preston, S. D., R. B. Alexander, G. E. Schwarz, and C. G. Crawford. 2011. Factors Affecting Stream Nutrient Loads: A Synthesis of Regional SPARROW Model Results for the Continental United States. JAWRA Journal of the American Water Resources Association **47**:891-915.

Pulwarty, R. S., and T. S. Melis. 2001. Climate extremes and adaptive management on the Colorado River: Lessons from the 1997-1998 ENSO event. Journal of Environmental Management **63**:307-324.

Qian, S. S., T. F. Cuffney, I. Alameddine, G. McMahon, and K. H. Reckhow. 2010. On the application of multilevel modeling in environmental and ecological studies. Ecology **91**:355-361.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Radcliffe, D. E., J. Freer, and O. Schoumans. 2009. Diffuse Phosphorus Models in the United States and Europe: Their Usages, Scales, and Uncertainties. Journal of Environmental Quality **38**:1956-1967.

Reckhow, K. 2003. On the need for uncertainty assessment in TMDL modeling and implementation. Journal of Water Resources Planning and Management-Asce **129**:245-246.

Reckhow, K. H., M. N. Beaulac, and J. T. Simpson. 1980. Modeling phosphorus loading and lake response under uncertainty: A manual and compilation of export coefficients. Washington D.C.

Regan, H. M., M. Colyvan, and M. A. Burgman. 2002. A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecological Applications **12**:618-628.

Richards, R. P., D. B. Baker, and J. P. Crumrine. 2009. Improved water quality in Ohio tributaries to Lake Erie: A consequence of conservation practices. Journal of Soil and Water Conservation **64**:200-211.

Riggs, D. S., J. A. Guarnieri, and S. Addelman. 1978. Fitting straight lines when both variables are subject to error. Life Sciences **22**:1305-1360.

Rodriguez, H. G., J. Popp, C. Maringanti, and I. Chaubey. 2011. Selection and placement of best management practices used to reduce water quality degradation in Lincoln Lake watershed. Water Resources Research **47**.

Said, A. 2006. The implementation of a Bayesian network for watershed management decisions. Water Resources Management **20**:591-605.

Scheffer, M., and S. R. Carpenter. 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends in Ecology & Evolution **18**:648-656.

Schindler, D. W., R. E. Hecky, D. L. Findlay, M. P. Stainton, B. R. Parker, M. J. Paterson, K. G. Beaty, M. Lyng, and S. E. M. Kasian. 2008. Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. Proceedings of the National Academy of Sciences **105**:11254-11258.

Schmelter, M. L., and D. K. Stevens. 2013. Traditional and Bayesian Statistical Models in Fluvial Sediment Transport. Journal of Hydraulic Engineering-Asce **139**:336-340.

Schreiber, E. S. G., A. R. Bearlin, S. J. Nicol, and C. R. Todd. 2004. Adaptive management: a synthesis of current understanding and effective application. Ecological Management & Restoration **5**:177-182.

Schueler, T., D. Hirschman, M. Novotney, and J. Zielinski. 2007. Urban Stormwater Retrofit Practices. Ellicott City MD.

Sharpley, A., and H. Tunney. 2000. Phosphorus research strategies to meet agricultural and environmental challenges of the 21st century. Journal of Environmental Quality **29**:176-181.

Shirmohammadi, A., I. Chaubey, R. D. Harmel, D. D. Bosch, R. Munoz-Carpena, C. Dharmasri, A. Sexton, M. Arabi, M. L. Wolfe, J. Frankenberger, C. Graff, and T. M. Sohrabi. 2006. Uncertainty in TMDL models. Transactions of the Asabe **49**:1033-1049.

Shrestha, S., F. Kazama, L. T. H. Newham, M. S. Babel, R. S. Clemente, H. Ishidaira, K. Nishida, and Y. Sakamoto. 2008. Catchment scale modelling of point source and non-point source pollution loads using pollutant export coefficients determined from long-term in-stream monitoring data. Journal of Hydro-Environment Research **2**:134-147.

Slovic, P. 1987. Perception of risk. Science **236**:280-285.

Slovic, P. 1995. The construction of preference. American Psychologist **50**:364-371.

Slovic, P., B. Fischhoff, and S. Lichtenstein. 1977. Behavioral decision-theory. Annual Review of Psychology **28**:1-39.

Smeltzer, E. 1997. A Phosphorus Budget, Model, and Load Reduction Strategy for Lake Champlain. Submitted to US Environmental Protection Agency Region I and Region II.

Smeltzer, E., F. Dunlap, and M. Simoneau. 2009. Lake Champlain Phosphorus Concentrations and Loading Rates, 1990-2008. Technical Report No. 57. Lake Champlain Basin Program, Grand Isle, VT.

Smeltzer, E., A. D. Shambaugh, and P. Stangel. 2012. Environmental change in Lake Champlain revealed by long-term monitoring. Journal of Great Lakes Research **38, Supplement 1**:6-18.

Smeltzer, E., and M. Simoneau. 2008. Phosphorus Loading to Missisquoi bay from Sub-Basins in Vermont and Quebec, 2002-2005. Vermont Agency of Natural Resources and Ministere du Developpement durable, de l'Environnement at des Parcs.

Smil, V. 2000. Phosphorus in the environment: Natural flows and human interferences. Annual Review of Energy and the Environment **25**:53-88.

Smith, L., M. C. Watzin, and G. Druschel. 2011. Relating sediment phosphorus mobility to seasonal and diel redox fluctuations at the sediment-water interface in a eutrophic freshwater lake. Limnology and Oceanography **56**:2251-2264.

Snell, A., and B. Brower. 2011. New York Agricultural Best Management Practice Tracking. Technical Report No. 62. Lake Champlain Basin Program, Grand Isle, VT.

Sondergaard, M., R. Bjerring, and E. Jeppesen. 2013. Persistent internal phosphorus loading during summer in shallow eutrophic lakes. Hydrobiologia **710**:95-107.

Stankey, G. H., R. N. Clark, and B. T. Bormann. 2005. Adaptive Management of Natural Resources: Theory, Concepts, and Management Institutions. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station,, Portland, OR.

State of Vermont. 2014. Vermont Lake Champlain Phosphorus TMDL Phase 1 Implementation Plan. Prepared for US EPA Region 1 by the Vermont Agency of Natural Resources, Vermont Agency of Agriculture, Food and Markets, & Vermont Agency of Transportation. Montpelier, VT, available at http://www.watershedmanagement.vt.gov/erp/champlain/. Accessed 2/18/14.
Stehman, S. V. 2013. Estimating area from an accuracy assessment error matrix. Remote Sensing of Environment **132**:202-211.
Sterner, R. W. 2008. On the Phosphorus Limitation Paradigm for Lakes. International Review of Hydrobiology **93**:433-445.
Stickney, M. 2008. Building bridges, fording streams, reaching agreement in the Lake Champlain basin: Alternatives to legislation and regulation rooted in citizen and science-based approaches to inspire watershed protection. Water Sa **34**:468-475.
Stickney, M., C. Hickey, and R. Hoerr. 2001. Lake Champlain Basin Program: Working together today for tomorrow. Lakes & Reservoirs: Research & Management **6**:217-223.
Tetra Tech. 2013a. Lake Champlain Basin SWAT Model Configuration, Calibration and Validation. Prepared for EPA Region 1 - New England, Boston MA.
Tetra Tech. 2013b. Lake Champlain Basin SWAT Model Configuration, Calibration, and Validation. Prepared for U.S. EPA Region 1 - New England. Boston MA. Available from http://www.epa.gov/region1/eco/tmdl/lakechamplain.html.
Tetra Tech. 2013c. Lake Champlain BATHTUB Model Calibration Report. Prepared for U.S. EPA Region 1 - New England. Boston MA. Available from http://www.epa.gov/region1/eco/tmdl/lakechamplain.html.
Troy, A., D. Wang, D. Capen, J. O'Neil-Dunne, and S. MacFadden. 2007. Updating the Lake Champlain Basin Land Use Data to Improve Prediction of Phosphorus Loading. Technical Report No. 54. Lake Champlain Basin Program, Grand Isle, VT.
Tversky, A., and D. Kahneman. 1974. Judgement under Uncertainty: Heuristics and Biases. Science **185**:1124-1131.
U.S. Fish and Wildlife Service. 2011. Adaptive Harvest Management: 2011 Hunting Season. U.S. Deptartment of the Interior, Washington D.C. Available online at http://flyways.us/sites/default/files/uploads/2011_ahm_report.pdf.
United States Department of Agriculture. 2009a. 2007 Census of Agriculture: New York State and County Data. AC-07-A-32.*in* National Agricultural Statistics Service, editor. USDA.
United States Department of Agriculture. 2009b. 2007 Census of Agriculture: Vermont State and County Data. AC-07-A-45.*in* National Agricultural Statistics Service, editor. USDA.
Varis, O. 1997. Bayesian decision analysis for environmental and resource management. Environmental Modelling & Software **12**:177-185.
Varis, O., and S. Kuikka. 1999. Learning Bayesian decision analysis by doing: lessons from environmental and natural resources management. Ecological Modelling **119**:177-195.
Vermont Agency of Natural Resources. 2010. A Framework for Remediation of Vermont's Stormwater-Impaired Waters. Vermont Department of Environmental Conservation, Waterbury, VT.
Vermont Agency of Natural Resources. 2013. Reported sewage overflows since 2007. URL: http://www.watershedmanagement.vt.gov/ww/om/Overflows/ReportedSewageOverflowsSince2007.pdf.

Vermont Agency of Natural Resources and Vermont Agency of Agriculture Food and Markets (VTANR & VTAAFM). 2011. Vermont Clean and Clear Action Plan 2010 Annual Report.

Vermont Center for Geographic Information. 2006. Vermont Hydrologically Corrected Digital Elevation Model.*in* VCGI, editor. VCGI, Waterbury, VT.

von Winterfeldt, D., and W. Edwards. 1986. Decision Analysis and Behavioral Research. Cambridge University Press, London.

VT DEC. 2014. Submitted Public Comments on the Vermont Lake Champlain Phosphorus TMDL Phase 1 Implementation Plan., Availale at: http://www.watershedmanagement.vt.gov/erp/champlain/cmnts-tmdlphase1.htm. Last accessed Feb 22, 2015.

Walling, D. E. 1999. Linking land use, erosion and sediment yields in river basins. Hydrobiologia **410**:223-240.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society **24**:706-723.

Walters, C. J. 1986. Adaptive Management of Renewable Resources. Macmillan Publishing Company, New York, NY.

Walters, C. J. 1997. Challenges in adaptive management of riparian and coastal ecosystems. Conservation Ecology **1**:1. [online] URL: http://www.consecol.org/vol1/iss2/art1/.

Walters, C. J. 2007. Is adaptive management helping to solve fisheries problems? Ambio **36**:304-307.

Watzin, M. C., S. Fuller, L. Bronson, R. Gorney, and L. Schuster. 2011. Monitoring and Evaluation of Cyanobacteria in Lake Champlain. Technical Report No. 61. Grand Isle, VT.

Watzin, M. C., R. L. Smyth, E. A. Cassell, W. C. Hession, R. E. Manning, and D. Wang. 2005. Ecosystem Indicators and an Environmental Score Card for the Lake Champlain Basin Program. Technical Report No. 46. Lake Champlain Basin Program, Grand Isle, VT.

Weller, C. M., M. C. Watzin, and D. Wang. 1996. Role of wetlands in reducing phosphorus loading to surface water in eight watersheds in the Lake Champlain Basin. Environmental Management **20**:731-739.

Wemple, B. 2013. Assessing the Effects of Unpaved Roads on Lake Champlain Water Quality. Technical Report No. 74. Lake Champlain Basin Program, Grand Isle, VT.

Wichelns, D. 2010. Embracing Uncertainty to Improve Water Management, with Examples from Seven River Basins. International Journal of Water Resources Development **26**:495-508.

Wickham, H. 2009. ggplot2: elegant graphics for data analysis. Springer, New York.

Wickham, J. D., S. V. Stehman, J. A. Fry, J. H. Smith, and C. G. Homer. 2010. Thematic accuracy of the NLCD 2001 land cover for the conterminous United States. Remote Sensing of Environment **114**:1286-1296.

Wickham, J. D., S. V. Stehman, L. Gass, J. Dewitz, J. A. Fry, and T. G. Wade. 2013. Accuracy assessment of NLCD 2006 land cover and impervious surface. Remote Sensing of Environment **130**:294-304.

Williams, B. K. 1996. Adaptive optimization of renewable natural resources: Solution algorithms and a computer program. Ecological Modelling **93**:101-111.

Williams, B. K. 2011a. Adaptive management of natural resources-framework and issues. Journal of Environmental Management **92**:1346-1353.
Williams, B. K. 2011b. Passive and active adaptive management: Approaches and an example. Journal of Environmental Management **92**:1371-1378.
Williams, B. K., and E. D. Brown. 2012. Adaptive Management: The U.S. Department of the Interior Applications Guide. Department of the Interior Adaptive Management Working Group, Washington DC.
Williams, B. K., M. J. Eaton, and D. R. Breininger. 2011. Adaptive resource management and the value of information. Ecological Modelling **222**:3429-3436.
Winchell, M., D. W. Meals, S. Folle, J. Moore, D. C. Braun, C. DeLeo, and K. Budreski. 2011. Identification of Critical Source Areas of Phosphorus Within the Vermont Sector of the Missisquoi Bay Basin. Technical Report No. 63b. Lake Champlain Basin Program, Grand Isle, VT.
Winsten, J. R. 2004. Developing and Assessing Policy Options for Reducing Phosphorus Loading in Lake Champlain. Technical Report No. 47. Lake Champlain Basin Program, Grand Isle, VT.
Wolfslehner, B., and H. Vacik. 2011. Mapping indicator models: From intuitive problem structuring to quantified decision-making in sustainable forest management. Ecological Indicators **11**:274-283.
Zobrist, J., and P. Reichert. 2006. Bayesian estimation of export coefficients from diffuse and point sources in Swiss watersheds. Journal of Hydrology **329**:207-223.
Zuzel, J. F., and J. L. Pikul. 1987. Infiltration into a seasonally frozen agricultural soil. Journal of Soil and Water Conservation **42**:447-450.

APPENDIX A — PHOSPHORUS INDICATOR TABLES

Bouquet & Ausable Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping									1,2	
b. Alternative manure spreading methods									1,2	
c. Conservation tillage									1,2	
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)									3	
Percent of farms operating within 5% of whole-farm P balance										
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	6/7	7/7	100%	0.01	0.01	$140,000	$730,000	$3,154	7,8,32	1
b. Silage leachate treatment (practices/farms)	1/7	7/7	100%	0.62	0.62	$270,000	$580,500	$47	7,8,32	1
c. Barnyard runoff treatment (practices/farms)	6/7	1/7	100%	0.02	0.02	$15,000	$111,000	$244	7,8,32	1
d. Milkhouse waste treatment (practices/farms)	5/7	7/7	100%	0.21	0.21	$33,250	$93,500	$23	7,8,32	1
Percent of farm inspections identifying substantial violations of relevant agricultural regulation										
Percent of perennial stream miles where livestock have uncontrolled access to the stream										
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit										
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit										
Percent of all permitted operational stormwater sites in substantial compliance with their permit										
Percent of municipalities with storm sewer systems that have completed IDDE projects										
Percent of impervious area that is under stormwater management	0.59%		100%		2.3	$291,018,052	$291,051,302	$6,224		
Number of combined sewer overflows remaining in the Lake Champlain Basin	0		0		0	$0	$0			
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment										
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.										
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin										
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.										
Percent of towns participating in the Better Backroads Program (or equivalent program)	92%									
Percent of towns that have completed road erosion needs inventories and capital budget plans	92%									
Percent of priority erosion control projects identified in road erosion needs inventories that are completed										
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored										
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs										
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances										
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans										

Bouquet & Ausable Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (R)	88%	100%	100%	0.01	0.01					
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	NA	a. 100% b. 75%	a. 100% b. 100%							
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre										
Ratio of imported P / exported P on agricultural lands										
avg. mt/yr P loss from cropland (including hay)	4.0									
avg. mt/yr P loss from farmsteads	0.2									
Ratio of imported P / exported P on urban lands										
avg. mt/yr P loss from urban areas	4.9									
avg. mt/yr P loss from road network										
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	1.94%									
b. hay, pasture, lawn	7.42%									
c. impervious surface	1.73%									
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)										
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	2.52									
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	93									
5 year avg. total tributary P loads (2007-2011) (mt/y)	95									
5-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.381									

Grand Isle & Direct Drainage Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction (mt/yr)	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping	1.74%	11.10%	54.97%	1.0	5.5	$415,087	$8,301,730	$75	1,2,5,17	2
b. Alternative manure spreading methods	0.00%	16.40%	100.0%	1.7	10.3	$1,027,627	$20,552,540	$99	1,2,5,18	2
c. Conservation tillage	2.63%	0.47%	54.97%	0.0	11.6	$413,273	$8,265,456	$36	1,2,5,17	2
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)	64.70%	75.00%	100%	3.0	10.2				3	3
Percent of farms operating within 5% of whole-farm P balance			100%							
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	L - NA M - 28/17	L - NA M - 28/17	100%	0.0	0.0	$0	$0	NA	5,6,7	1
b. Silage leachate treatment (practices/farms)	L - NA M - 5/17	L - NA M - 17/17	100%	0.7	0.7	$540,000	$1,120,500	$84	5,6,8	1
c. Barnyard runoff treatment (practices/farms)	L - NA M - 20/17	L - NA M - 20/17	100%	0.0	0.0	$0	$0	NA	5,6,7	1
d. Milkhouse waste treatment (practices/farms)	L - NA M - 1/17	L - NA M - 17/17	100%	0.9	0.9	$266,000	$559,000	$31	5,6,8	1
Percent of farm inspections identifying substantial violations of relevant agricultural regulation	5.00%		0%		0.0				5	3
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit	46%	NA	100%	NA	0.0			NA	5	3
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit	90%	NA	100%	NA	0.0			NA	5	3
Percent of all permitted operational stormwater sites in substantial compliance with their permit	85%	NA	100%	NA	0.0			NA	5	3
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	16.26%	NA	100%	NA	2.8	$88,345,542	$88,378,792	$1,593	9,11,19,20,21	4
Number of combined sewer overflows remaining in the Lake Champlain Basin	1		0		0.03	$25,054,719	$25,054,719	$41,758	12,13,22,30	12
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.	23%		100%						5	
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin	10.10%		40%						23	
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.	17%		0%						24	3
Percent of towns participating in the Better Backroads Program (or equivalent program)	60%		100%	0.0	0.0			NA	5	10
Percent of towns that have completed road erosion needs inventories and capital budget plans	16%		100%	0.0	0.0			NA	5	3, 10
Percent of priority erosion control projects identified in road erosion needs inventories that are completed	57%		100%		0.5	$646,707	$1,875,450	$177	5,15,16	3, 8, 14
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs	75%		100%						25	
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances	0%		100%						5	
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Grand Isle & Direct Drainage Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction (mt/yr)	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (3-yr avg)	100%	100%	100%	0.0	0.0				26	
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	(a) 100% (b) 0%	a. 100% b. 75%	a. 100% b. 100%						27	
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre	0.58								3	3
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	49.2		TBD						2	
avg. mt/yr P loss from farmsteads	0.1		TBD						2	
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	6.9		TBD						2	
avg. mt/yr P loss from road network	1.9		TBD						2	
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	21.64%								2	
b. hay, pasture, lawn	26.25%								2	
c. impervious surface	3.61%								2	
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)	71%	50%	30%							8
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	21.84		TBD						28,29	
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	196		TBD						28,29	
5 year avg. total tributary P loads (2007-2011) (mt/y)	218		TBD						28,29	
6-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)										

Lamoille Basin	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping	2.94%	11%	32.0%	0.6	2.0	$156,208	$3,124,159	$79	1,2,5,17	2
b. Alternative manure spreading methods	0.00%	16.40%	100.0%	0.8	4.8	$1,053,420	$21,068,401	$221	1,2,5,18	2
c. Conservation tillage	0.21%	0.47%	32.0%	0.0	3.2	$160,997	$3,219,949	$50	1,2,5,17	2
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)	64.70%	75.00%	100%	1.4	4.7				3	3
Percent of farms operating within 5% of whole-farm P balance			100%							
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	L - NA M - 19/10	L - NA M - 19/10	100%	0.00	0.00	$0	$0	NA	5,6,7	1
b. Silage leachate treatment (practices/farms)	L - NA M - 4/10	L - NA M - 10/10	100%	0.33	0.33	$270,000	$580,500	$87	5,6,8	1
c. Barnyard runoff treatment (practices/farms)	L - NA M - 11/10	L - NA M - 11/10	100%	0.00	0.00	$0	$0	NA	5,6,7	1
d. Milkhouse waste treatment (practices/farms)	L - NA M - 0/10	L - NA M - 10/10	100%	0.56	0.56	$166,250	$359,500	$32	5,6,8	1
Percent of farm inspections identifying substantial violations of relevant agricultural regulation	5.00%		0%	0.0	0.0				5	3, 10
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit	46%	NA	100%	NA	0.0			NA	5	3, 10
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit	90%	NA	100%	NA	0.0			NA	5	3, 10
Percent of all permitted operational stormwater sites in substantial compliance with their permit	85%	NA	100%	NA	0.0			NA	5	3, 10
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	5.27%	NA	100%	NA	4.9	$237,065,169	$237,098,419	$2,437	9,11,19,20,21	4
Number of combined sewer overflows remaining in the Lake Champlain Basin	0	0	0	0	0	$0		$0	12,13,22,30	
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.	23%		100%						5	
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin	12.76%		40%						23	
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.	17%		0%						24	3
Percent of towns participating in the Better Backroads Program (or equivalent program)	85%		100%	0	0			NA	5	10
Percent of towns that have completed road erosion needs inventories and capital budget plans	16%		100%	0	0			NA	5	3, 10
Percent of priority erosion control projects identified in road erosion needs inventories that are completed	57%		100%		3.3	$1,397,313	$4,052,207	$61	5,15,16	3, 14
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs	60%		100%						25	
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances	5%		100%						5	
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Lamoille Basin	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (R)	88%	100%	100%	0.0	0.0				26	
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	(a) 100% (b) 10%	a. 100% b. 75%	a. 100% b. 100%						27	
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre	0.49								3	3
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	22.7		TBD						2	
avg. mt/yr P loss from farmsteads	0.5		TBD						2	
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	10.7		TBD						2	
avg. mt/yr P loss from road network	11.8		TBD						2	
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	2.77%								2	
b. hay, pasture, lawn	10.45%								2	
c. impervious surface	1.99%								2	
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)	75%	50%	30%							
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	1.31		TBD						28,29	
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	68		TBD						28,29	
5 year avg. total tributary P loads (2007-2011) (mt/y)	69		TBD						28,29	
6-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.347								28,29	

Missisquoi Basin	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping	1.06%	11.10%	54.44%	2.7	14.4	$1,673,455	$33,469,106.87	$116	1,2,4,5,17	2
b. Alternative manure spreading methods	17.99%	37.21%	100.00%	3.6	15.5	$3,595,166	$71,903,322.13	$231	1,2,4,5,17	2
c. Conservation tillage	16.40%	21.30%	54.44%	2.0	15.4	$1,625,611	$32,512,220.77	$105	1,2,4,5,17	2
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)	83.99%	83.99%	100%	0.0	8.5				3,4,10	3
Percent of farms operating within 5% of whole-farm P balance			100%							
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	L - 18/7 M - 25/30	L - 18/7 M - 30/30	100%	0.05	0.05	$700,000	$1,850,000	$685	5,6,7,10	1,8
b. Silage leachate treatment (practices/farms)	L - 5/7 M - 17/30	L - 7/7 M - 30/30	100%	1.02	1.02	$675,000	$1,390,500	$33	5,6,8,10	1,8
c. Barnyard runoff treatment (practices/farms)	L - 4/7 M - 29/30	L - 7/7 M - 30/30	100%	0.02	0.02	$60,000	$201,000	$175	5,6,7,10	1,8
d. Milkhouse waste treatment (practices/farms)	L - 2/7 M - 2/30	L - 7/7 M - 30/30	100%	2.31	2.31	$548,625	$1,124,250	$12	5,6,8,10	1,8
Percent of farm inspections identifying substantial violations of relevant agricultural regulation	9.00%		0%		0.0				5, 10	3
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit	46%	NA	100%	NA	0.0			NA	5	3,8,10
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit	90%	NA	100%	NA	0.0			NA	5	3,8,10
Percent of all permitted operational stormwater sites in substantial compliance with their permit	85%	NA	100%	NA	0.0			NA	5	3,8,10
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	1.85%	NA	100%	NA	7.2	$181,222,179	$182,206,379	$1,264	9,11,19,20,21	4,8
Number of combined sewer overflows remaining in the Lake Champlain Basin	22		0		0.13	$167,187,649	$167,187,649	$64,303	10,12,13,22,30	10, 12
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.	67%		100%						5,10	
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin	7.84%		40%						23	
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.	17%		0%						24	3
Percent of towns participating in the Better Backroads Program (or equivalent program)	19%		100%	0.0	0.0			NA	5	10
Percent of towns that have completed road erosion needs inventories and capital budget plans	16%		100%	0.0	0.0			NA	5	3, 10
Percent of priority erosion control projects identified in road erosion needs inventories that are completed	57%		100%		3.3	$10,597,440	$30,732,575	$462	5,15,16	3, 8, 14
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs	94%		100%						10,25	
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances	2%		100%						5	8
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Missisquoi Basin	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (R)	86%	100%	100%	0.27	0.27				26	
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	(a) 100% (b) 0%	a. 100% b. 75%	a. 100% b. 100%						27	
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre	0.7								3	3,8
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	90.2		TBD						2	
avg. mt/yr P loss from farmsteads	0.9		TBD						2	
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	15.3		TBD						2	
avg. mt/yr P loss from road network	11.9		TBD						2	
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	11.50%								2	
b. hay, pasture, lawn	14.64%								2	
c. impervious surface	1.63%								2	
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)	55%	50%	30%							8
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	2.15		TBD						28,29	
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	261		TBD						28,29	
5 year avg. total tributary P loads (2007-2011) (mt/y)	263		TBD						28,29	
5-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.386								28,29	

Otter & Lewis Creek Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per Expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping	0.91%	11.10%	33.30%	2.3	7.2	$528,855	$10,577,100	$73	1,2,5,17	2
b. Alternative manure spreading methods	1.77%	16.40%	100.00%	2.3	15.4	$3,327,736	$66,554,716	$217	1,2,5,18	2
c. Conservation tillage	5.06%	0.47%	33.30%	0.0	9.5	$504,769	$10,095,378	$53	1,2,5,17	2
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)	64.70%	75.00%	100%	4.5	15.5				3	3
Percent of farms operating within 5% of whole-farm P balance			100%							
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	L - 6/4 M - 49/35	L - 6/4 M - 49/35	100%	0.0	0.0	$0	$0	$0	5,6,7	1
b. Silage leachate treatment (practices/farms)	L - 1/4 M - 13/35	L - 3/4 M - 35/35	100%	1.7	1.7	$1,125,000	$2,290,500	$68	5,6,8	1
c. Barnyard runoff treatment (practices/farms)	L - 2/4 M - 37/35	L - 4/4 M - 37/35	100%	0.0	0.0	$30,000	$141,000	$865	5,6,7	1
d. Milkhouse waste treatment (practices/farms)	L - 0/4 M - 0/35	L - 4/4 M - 35/35	100%	2.6	2.6	$648,375	$1,323,750	$26	5,6,8	1
Percent of farm inspections identifying substantial violations of relevant agricultural regulation	5.00%		0%		0.0				5	3
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit	46%	NA	100%	NA	0.0			NA	5	3
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit	90%	NA	100%	NA	0.0			NA	5	3
Percent of all permitted operational stormwater sites in substantial compliance with their permit	85%	NA	100%	NA	0.0			NA	5	3
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	5.60%	NA	100%	NA	7.9	$310,684,961	$310,718,211	$1,959	9,11,19,20,21	4
Number of combined sewer overflows remaining in the Lake Champlain Basin	7		0		0.16	$17,453,006	$17,453,006	$5,454	12,13,22,30	12
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.	23%		100%						5	
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin	11.50%		40%						23	
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.	17%		0%						24	3
Percent of towns participating in the Better Backroads Program (or equivalent program)	71%		100%	0.0	0.0			NA	5	10
Percent of towns that have completed road erosion needs inventories and capital budget plans	16%		100%	0.0	0.0			NA	5	3, 10
Percent of priority erosion control projects identified in road erosion needs inventories that are completed	57%		100%		3.9	$1,195,468	$3,466,856	$44	5,15,16	3, 8, 14
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs	62%		100%						25	
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances	9%		100%						5	
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Otter & Lewis Creek Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per Expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (R)	100%	100%	100%	0.0	0.0				26	
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	(a) 43% (b) 0%	a. 100% b. 75%	a. 100% b. 100%						27	
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre	0.489								3	3
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	74.5		TBD						2	
avg. mt/yr P loss from farmsteads	1.3		TBD						2	
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	17.5		TBD						2	
avg. mt/yr P loss from road network	14.1		TBD						2	
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	6.42%								2	
b. hay, pasture, lawn	19.18%								2	
c. impervious surface	1.85%								2	
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)	43%	50%	30%							
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	3.89		TBD						28,29	
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	172		TBD						28,29	
5 year avg. total tributary P loads (2007-2011) (mt/y)	176		TBD						28,29	
5-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.385								28,29	

Poultney & Mettowee Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping	0.88%	11.10%	33.00%	0.9	2.9	$463,982	$9,279,634	$161	1,2,5,17	2
b. Alternative manure spreading methods	0.47%	16.40%	100.00%	1.0	6.2	$2,283,407	$45,668,148	$365	1,2,5,18	2
c. Conservation tillage	2.65%	0.47%	33.00%	0.0	4.1	$460,265	$9,205,309	$113	1,2,5,17	2
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)	64.70%	75.00%	100%	1.8	6.2				3	3
Percent of farms operating within 5% of whole-farm P balance			100%							
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	47/20	47/20	100%	0.0	0.0	$0	$0	NA	5,6,7,32	1
b. Silage leachate treatment (practices/farms)	12/20	20/20	100%	0.8	0.8	$360,000	$760,500	$46	5,6,8,32	1
c. Barnyard runoff treatment (practices/farms)	54/20	54/20	100%	0.0	0.0	$0	$0	NA	5,6,7,32	1
d. Milkhouse waste treatment (practices/farms)	8/20	20/20	100%	1.2	1.2	$199,500	$426,000	$17	5,6,8,32	1
Percent of farm inspections identifying substantial violations of relevant agricultural regulation	5.00%		0%		0.0				5	3
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit	46%	NA	100%	NA	0.0			NA	5	3
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit	90%	NA	100%	NA	0.0			NA	5	3
Percent of all permitted operational stormwater sites in substantial compliance with their permit	85%	NA	100%	NA	0.0			NA	5	3
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	1.36%	NA	100%	NA	6.2	$440,631,743	$440,664,993	$3,580	9,11,19,20,21	4,5,8
Number of combined sewer overflows remaining in the Lake Champlain Basin	2		0		0.03	$18,574,384	$18,574,384	$30,957	12,13,22,30	12
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.	23%		100%						5	
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin	11.80%		40%						23	
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.	17%		0%						24	3
Percent of towns participating in the Better Backroads Program (or equivalent program)	94%		100%	0.0	0.0			NA	5	10
Percent of towns that have completed road erosion needs inventories and capital budget plans	16%		100%	0.0	0.0			NA	5	3, 10
Percent of priority erosion control projects identified in road erosion needs inventories that are completed	57%		100%		1.4	$424,973	$1,232,420	$43	5,15,16	3, 8, 14
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs	94%		100%						25	
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances	3%		100%						5	
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Poultney & Mettowee Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (R)	100%	100%	100%	0.0	0.0				26	
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	(a) 90% (b) 20%	a. 100% b. 75%	a. 100% b. 100%						27	
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre									3	3
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	29.9		TBD						2	
avg. mt/yr P loss from farmsteads	0.3		TBD						2	
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	13		TBD						2	
avg. mt/yr P loss from road network	5.1		TBD						2	
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	3.66%								2	
b. hay, pasture, lawn	13.81%								2	
c. impervious surface	2.18%								2	
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)	41%	50%	30%							8
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	1.48		TBD						28,29	
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	185		TBD						28,29	
5 year avg. total tributary P loads (2007-2011) (mt/y)	187		TBD						28,29	
5-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.378								28,29	

Saranac & Chazy Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Cost of reaching ultimate acceptable level ($)	Real cost to reach ultimate level (20-year) ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of agricultural land under enhanced land management for:										
a. Cover cropping										
b. Alternative manure spreading methods										
c. Conservation tillage										
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)										
Percent of farms operating within 5% of whole-farm P balance										
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	16/13	16/13	100%	0.00	0.00	$0	$0	$0	7,8,32	1
b. Silage leachate treatment (practices/farms)	7/13	13/13	100%	0.62	0.62	$270,000	$580,500	$47	7,8,32	
c. Barnyard runoff treatment (practices/farms)	5/13	13/13	100%	0.10	0.12	$120,000	$321,000	$137	7,8,32	
d. Milkhouse waste treatment (practices/farms)	6/13	13/13	100%	0.72	0.72	$116,375	$259,750	$18	7,8,32	
Percent of farm inspections identifying substantial violations of relevant agricultural regulation			0%							
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit			100%							
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit			100%							
Percent of all permitted operational stormwater sites in substantial compliance with their permit			100%							
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	2.10%		100%		3.0	$343,180,686	$343,213,936	$5,706		
Number of combined sewer overflows remaining in the Lake Champlain Basin	11	5	0	0.07	0.15	$49,069,946	$49,069,946	$16,357		
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Number of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.			100%							
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin			40%							
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.			0%							
Percent of towns participating in the Better Backroads Program (or equivalent program)	53%		100%		0.0					
Percent of towns that have completed road erosion needs inventories and capital budget plans	53%		100%		0.0					
Percent of priority erosion control projects identified in road erosion needs inventories that are completed			100%							
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs			100%							
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances			100%							
Rolling 15 year cumulative totals for acres of identified priority wetlands (a) restored and (b) conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Saranac & Chazy Basins	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction	Expected ultimate P reduction	Cost of reaching ultimate acceptable level ($)	Real cost to reach ultimate level (20-year) ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations (R)	100%	100%	100%	0.0	0.0					
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system (P)	NA	a. 100% b. 75%	a. 100% b. 100%							
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre										
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	9.9									
avg. mt/yr P loss from farmsteads	0.4									
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	6.4									
avg. mt/yr P loss from road network										
Mean soil P level in cropland (includes annual and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	1.96%									
b. hay, pasture, lawn	8.84%									
c. impervious surface	2.22%									
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening) (R)		50%	30%							
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	4.26									
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	65									
5 year avg. total tributary P loads (2007-2011) (mt/y)	69									
5-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.535									

Winooski Basin	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction (mt/yr)	Expected ultimate P reduction (mt/yr)	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Agricultural Lands										
Percent of cropland (incl. hay) under enhanced land management for:										
a. Cover cropping	1.50%	11.10%	29.20%	0.8	2.4	$196,065	$3,921,297.16	$83	1,2,5,17	2
b. Alternative manure spreading methods	0.13%	16.40%	100.00%	1.0	6.0	$1,524,978	$30,499,569.33	$255	1,2,5,18	2
c. Conservation tillage	2.33%	0.47%	29.20%	0.0	3.4	$194,067	$3,881,335.06	$56	1,2,5,17	2
Percent of agricultural land acres managed under an approved Nutrient Management Plan, by farm type (LFO, MFO, SFO)	64.70%	75.00%	100%	1.7	5.9				3	3,9
Percent of farms operating within 5% of whole-farm P balance			100%							
Percent of regulated farms (LFOs/Large CAFOs & MFOs/Medium CAFOs) with regularly-maintained Best Management Practice structures, by farm type										
a. Manure storage (practices/farms)	L - 0/1 M - 23/11	L - 1/1 M - 23/11	100%	0.01	0.01	$140,000	$730,000	$3,476	5,6,7	1
b. Silage leachate treatment (practices/farms)	L - 0/1 M - 4/11	L - 1/1 M - 11/11	100%	0.54	0.54	$360,000	$760,500	$70	5,6,8	1
c. Barnyard runoff treatment (practices/farms)	L - 0/1 M - 28/11	L - 1/1 M - 28/11	100%	0.01	0.01	$15,000	$111,000	$419	5,6,7	1
d. Milkhouse waste treatment (practices/farms)	L - 0/1 M - 3/11	L - 1/1 M - 11/11	100%	0.60	0.60	$149,625	$326,250	$27	5,6,8	1
Percent of farm inspections identifying substantial violations of relevant agricultural regulation	5.00%		0%	0.0	0.0				5	3,7, 10
Percent of perennial stream miles where livestock have uncontrolled access to the stream			0%							
Developed Lands										
Percent of all permitted construction stormwater sites under the Construction General Permit in substantial compliance with the permit	46%	NA	100%	NA	0.0			NA	5	3, 10
Percent of all permitted construction stormwater sites with Individual Permits in substantial compliance with their permit	90%	NA	100%	NA	0.0			NA	5	3, 10
Percent of all permitted operational stormwater sites in substantial compliance with their permit	85%	NA	100%	NA	0.0			NA	5	3, 10
Percent of municipalities with storm sewer systems that have completed IDDE projects			100%							
Percent of impervious area that is under stormwater management	13.72%	NA	100%	NA	9.5	$484,363,023	$484,396,273	$2,543	9,11,19,20,21	4, 11
Number of combined sewer overflows remaining in the Lake Champlain Basin	10		0		0.08	$49,627,223	$49,627,223	$31,017	12,13,22,30	12
Percent of land area in stormwater impaired watersheds in need of treatment that is receiving treatment			100%							
Percentage of towns with good water quality protection provisions in town plans and zoning ordinances, including incorporation of Low Impact Development standards where appropriate.	23%		100%						5	3
Percent of tree canopy coverage within urban landscape zones in the Lake Champlain Basin	15.50%		40%						23	
Rural Lands/Backroads										
Percent of inspected sampling units within logging jobs in the Vermont and New York portions of the Lake Champlain Basin where harvesting operations have caused more than trace amounts of sediment to enter streams.	17%		0%						24	3
Percent of towns participating in the Better Backroads Program (or equivalent program)	72%		100%	0.0	0.0			NA	5	10
Percent of towns that have completed road erosion needs inventories and capital budget plans	16%		100%	0.0	0.0			NA	5	3, 10
Percent of priority erosion control projects identified in road erosion needs inventories that are completed	57%		100%		5.6	$1,536,233	$4,455,076	$39	5,15,16	3, 14
River, Floodplain, and Wetland Conservation & Restoration										
Percent of stream miles with perennial vegetated buffers in non-forested land use areas - differentiated by adjoining land use, buffer width class, vegetation type (woody, non-woody), programmatic coverage (e.g., CREP, WRP), and consistency with any regulatory standards that apply.										
Cumulative percent of river miles classified, as part of a statewide sediment regime departure analysis, to be unconfined, sediment transport reaches (i.e., incised reaches that should be depositional, and not under active management) for which floodplain access is either (a) actively or (b) passively restored			a. ? b. 80%							
Percent of towns having adopted Town and Bridge Standards in accordance with Act 110 that contain a suite of water quality based BMPs	66%		100%						25	3
Percent of Basin communities with adopted municipal Fluvial Erosion Hazard ordinances	19%		100%		0.0			NA	5	
Rolling 15 year cumulative totals for acres of identified priority wetlands restored or conserved										
Percentage of river corridor miles secured through easements for reaches of river identified as key sediment attenuation areas in completed geomorphic-based river corridor plans			100%							

Winooski Basin	Current State	Acceptable Level (short term)	Acceptable Level (ultimate)	Expected short-term P reduction (mt/yr)	Expected ultimate P reduction (mt/yr)	Initial Investments to reach ultimate acceptable level ($)	Real 20-year cost to reach ultimate level ($)	Expected Cost ($) per expected kg P	Data Sources	Footnotes
Wastewater										
Percent of facilities meeting their TMDL wasteload (VT & NY) or phosphorus (PQ) allocations	95%	100%	100%	0.77	0.77				26	
Percent of wastewater treatment facilities having an approved sewage spill prevention plan for (a) the treatment plant and (b) the collection system	(a) 100% (b) 35%	a. 100% b. 75%	a. 100% b. 100%						27	
Ecosystem Process & Ecosystem State Indicators:										
Median animal units per acre	0.45								3	3
Ratio of imported P / exported P on agricultural lands			1:1							
avg. mt/yr P loss from cropland (including hay)	28.5		TBD						2	
avg. mt/yr P loss from farmsteads	0.3		TBD						2	
Ratio of imported P / exported P on urban lands			1:1							
avg. mt/yr P loss from urban areas	23		TBD						2	
avg. mt/yr P loss from road network	20.2		TBD						2	
Mean soil P level in cropland (includes rotated and permanent hay)										
Mean soil P level in pastureland										
Best recent estimates for % of land in the following categories:										
a. annual crops	2.43%								2	
b. hay, pasture, lawn	11.10%								2	
c. impervious surface	2.98%								2	
Percent of river reaches in stream geomorphic assessment category II (incised and steepening) or III (incised and widening)	70%	50%	30%							
lbs P applied to developed lands										
5 year avg. wastewater phosphorus load (2007-2011) (mt/y)	8.26		TBD						28,29	
5 year avg. non-point phosphorus load (2007-2011) (mt/y)	244		TBD						28,29	
5 year avg. total tributary P loads (2007-2011) (mt/y)	252		TBD						28,29	
6-year Ratio of dissolved P : total P in tributary loads (2007-2012 conc.)	0.304								28,29	

Footnotes

1 These numbers represent the numbers of practices that have been cost-shared by the VT Agency of Ag. While most farms cost-share the structural practices listed, they are not required to, and therefore this may be an underestimate.

2 Ultimate acceptable levels are the proportion of total agricultural area that is in cropland (not including hay for cover crops and reduced tillage), according to the Land Use/Land Cover data layer developed by Tetra Tech. These estimates should be adjusted upward or downward using the CLU layer and the field selection model detailed in the accompanying report.

3 State-level or Lake Champlain Basin-level data, not watershed-specific data

4 Effectiveness Rate includes equally weighted means for all groups of treatment practices discussed in the Database.

5 % area treated is an underestimate, since it does not include any area for the NY side of the watershed - therefore, the estimate of ultimate reductions expected is an overestimate.

6 Refers only to the Vermont portion of the basin

7 No phosphorus reduction is associated with this indicator because the benefit to phosphorus loading as a result of changes in the value of this indicator are more directly captured in the values of another indicator in the table, or because implementation efforts under this indicator are not expected to achieve any real reductions in phosphorus loading.

8 Uses data from only the Vermont portion of the basin

9 The ultimate reduction estimate is a measure of bringing the remaining land-base under nutrient management, at the current rate of use of the plan, which is about 75% (i.e. farmers typically apply the recommendations to about 75% of their land in a particular year). The short term level reflects the commitments in OFA to ensure that the existing plans are utilized fully, without necessarily an increase in the land base under nutrient management

10 The phosphorus reduction for inspection programs and program enrollment is given as 0 here, according to the following logic. One common assumption of the reduction rates associated with BMPs are that those BMPs are maintained, and that (if structural) they are in place permanently, or (if an action) they are performed regularly. Clearly, these assumptions are not always valid. For the purposes of estimating reductions possible through managment programs in the LCB, reduction rates were applied to the practices themselves, with the assumption that inspection programs verify that the assumptions above are met. When they are not (i.e. when compliance rates are $< 100\%$), the realized reduction is less than the rate used in the estimation method. Bringing compliance rates to 100%, therefore, only serves to ensure that the realized reduction is closer to the estimated reduction, not to add any further reduction beyond the practices included in the estimation.

11 Does not include staff time to administer programs.

12 Includes current CSOs and recently abated CSOs that are under effectiveness monitoring to ensure compliance

13 Loading estimates from CSOs (and the reductions from eliminating them) were developed according to the following assumptions

14 The reduction estimate here is a measured reduction in sediment, not of total phosphorus. It is very likely this is an overestimate, which then underestimates the cost per kg P. There is essentially no data on P reductions from this sort of work.

Phosphorus Indicator Table Data Sources

1 Vermont Agency of Agriculture Farm Agronomic Program Database, delivered 01.10.2012

2 Tetra Tech. 2013. Lake Champlain Basin SWAT Model Configuration, Calibration and Validation. Prepared for EPA Region 1 - New England, Boston MA. Delivered 08.12.2013

3 H. Darby, P. Halteman, D. Heleba. Effectiveness of Nutrient Management Plans on Vermont Dairy Farms J. of Extension (in press).

4 2011 Canadian Agricultural Census Results, available at http://www.statcan.gc.ca/ca-ra2011/index-eng.htm

5 Vermont Agency of Natural Resources and Vermont Agency of Agriculture Food and Markets (VTANR & VTAAFM). 2012. Vermont Ecosystem Restoration Program 2011 Annual Report.

6 Vermont Agency of Agriculture Best Management Plan database, delievered with revisions 07.31.2013

7 Gitau, M. W., W. J. Gburek, and A. R. Jarrett. 2005. A tool for estimating best management practice effectiveness for phosphorus pollution control. Journal of Soil and Water Conservation 60:1-10.

8 Snell, A., and B. Brower. 2011. New York Agricultural Best Management Practice Tracking. Technical Report No. 62. Lake Champlain Basin Program, Grand Isle, VT.

9 Winer, R. 2000. National Pollutant Removal Performance Database for Stormwater Treatment Practices, 2nd Edition. Center for Watershed Protection, Ellicott City, MD.

10 Palov, T. 2012. Data Collection to Support the Development of an Adaptive Management Program at the Lake Champlain Basin Program

11 Schueler, T., D. Hirschman, M. Novotney, and J. Zielinski. 2007. Urban Stormwater Retrofit Practices. Ellicott City MD. Data from Hoyt (2007)

12 EPA. 2004. Report to Congress: Impacts and Control of CSOs and SSOs. EPA Office of Water, EPA 833-R-04-001, Washington DC.

13 EPA. 2007. The Long-Term Control Plan-EZ (LTCP-EZ) Template: A Planning Tool for CSO Control in Small Communities. EPA Office of Water, EPA-833-R-07-005, Washington DC.

14 Heath, G. R., G. M. Grant, and R. S. Seymour. 2004. Combined sewer separation not necessarily the answer to CSO control. Proceedings of the Water Environment Federation 2004:414-435.

15 LCLGRPB. 2012. Lake Champlain watershed water quality management planning: roadside erosion assessment and inventory. Lake Champlain-Lake George Regional Planning Board, Lake George, NY.

16 Turton, D. J., M. D. Smolen, and E. Stebler. 2009. Effectiveness of BMPS in Reducing Sediment From Unpaved Roads in the Stillwater Creek, Oklahoma Watershed. JAWRA Journal of the American Water Resources Association 45:1343-1351.

17 Michaud, A. R., I. Beaudin, J. Deslandes, F. Bonn, and C. A. Madramootoo. 2007. SWAT-predicted influence of different landscape and cropping system alterations on phosphorus mobility within the Pike River watershed of south-western Quebec. Canadian Journal of Soil Science 87:329-344.

18 Sedorovich, D. M., C. A. Rotz, P. A. Vadas, and R. D. Harmel. 2007. Simulating management effects on phosphorus loss from farming systems. Transactions of the Asabe 50:1443-1453.

19 Vermont Agency of Natural Resources. Permitted Impervious surface by basin and permit type. Delivered 10.25.2012

20 O'Neil-Dunne, J. 2013. Mapping Impervious Surfaces in the Lake Champlain Basin. Technical Report No. 76. Lake Champlain Basin Program, Grand Isle, VT.

21 Bissonnette Environmental Solutions, L. 2010. Task 1: Urban Stormwater Runoff Preliminary Needs Assessment Technical Memorandum. The Puget Sound Partnership, Olympia, WA. Available at http://www.psp.wa.gov/downloads/Stormwater/FinalUrbanStormwaterTechMemo20100930.pdf.

22 Vermont Agency of Natural Resources. Combined sewer overflow status in the Lake Champlain Basin, Delivered 09.28.2012

23 Vermont Agency of Forest, Parks, and Recreation. Urban Tree Canopy Assessment. Data delivered 03.07.2013

24 Ferrare, K. A., and P. K. Barten. 2006. Best Management Practice (BMP) Protocol: Monitoring Implementation and Effectiveness for Protection of Water Resources, Data Summary for Vermont. UMass Amherst Department of Natural Resources Conservation.

25 Vermont Agency of Natural Resources. Act 110 Codes and Standards Status by town. Data delivered 09.26.2012

26 Vermont Agency of Natural Resources. Wastewater Treatment Loading tables. Data delivered 10.01.2012

27 Vermont Agency of Natural Resources. Wastewater Treatment Facility and Collection System Spill Prevention Plan development status. Data delivered 09.26.2012

28 Vermont Agency of Natural Resources. Lake Champlain Long Term Water Quality and Biological Monitoring Project.

29 Hirsch, R. M., D. L. Moyer, and S. A. Archfield. 2010. Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay river inputs. JAWRA Journal of the American Water Resources Association 46:857-880.

30 Vermont Agency of Natural Resources. 2013. Reported sewage overflows since 2007. URL: http://www.watershedmanagement.vt.gov/ww/om/Overflows/ReportedSewageOverflowsSince2007.pdf.

31 Vermont Agency of Natural Resources. River Management Program Stream Geomorphic Assessment Results. Available from https://anrnode.anr.state.vt.us/SGA/finalReports.aspx

32 New York State Department of Environmental Conservation. Confined Animal Feeding Operation locations. Data delivered 07.11.2013

33 New York State Department of Environmental Conservation. Permitted Impervious Acreage. Data delivered 07.11.2013

34 New York State Department of Environmental Conservation. CSO abatement status. Data delivered 07.11.2013

APPENDIX B – INDICATOR BY INDICATOR CALCULATION NOTES

General Notes:

For the purposes of summing implementation data and the phosphorus loading data, the 18 gauged tributaries are divided into the 8 major basins as follows:

- "Missisquoi" includes the Missisquoi and Pike Rivers.
- "Winooski" includes the Winooski and La Platte Rivers
- "Lamoille" includes the Lamoille River and Mallet's Bay watersheds
- "Otter Creek" includes Otter, Little Otter, and Lewis Creeks
- "Poultney-Mettawee" includes the Poultney and Mettawee Rivers, and Putnam Creek.
- "Bouquet-Ausable" includes Bouquet, Little Ausable, Ausable, and Salmon River drainages.
- "Saranac-Chazy" includes the Saranac, Little Chazy, and Great Chazy Rivers.
- "Grand Isle/Direct" includes St. Albans Bay watershed, the Northeast Arm watershed, and the Isle La Motte watersheds.

Agricultural Lands:

% Ag land under enhanced management

Current State: Acreage of agricultural land was calculated from the Tetra Tech Land Use raster layer developed as part of the ongoing TMDL update, using tabulate areas by HUC 8 tool (Tetra Tech 2013a). Cover cropping acreages calculated from VT AAFM FAP program records, delivered 01/2012 by Nate Sands. "Cover cropping" includes the following practices: cover cropping, nurse cover crops, and cover crop seed incorporation practices. Conservation tillage includes conservation tillage, aeration tillage and cross-slope tillage. There is only one practice for manure methods. These data are an underestimate because there are other programs (e.g. through UVM Extension) that help farmers do cover cropping and conservation tillage, although there is overlap between those programs that is difficult to quantify. The difference might be as large as a factor of 2, but is likely much less. For Missisquoi Bay, the proportions for each practice are area-weighted. The data (from Stats Canada 2011 Agricultural Census) on cover cropping seems to indicate that little cover cropping occurs in the Quebec portion of the watershed. For conservation tillage, the proportion was calculated as the ratio of hectares with no-till and reduced tillage to the total land prepared for seeding. The average of these proportions was then multiplied by the area of cropland on either side of the border.

It would be a better estimate to use the acreage of land that is capable of supporting each of the practices (rather than total cropland), which could be done using the GIS soils layer and automated field selection routine (performed in ArcGIS ModelBuilder) developed by Philip Halteman in the fall of 2011. The process requires a current Common Land Unit (CLU) layer, which is held by VT AAFM or NRCS. The basic selection process selects fields capable of supporting these three practices based on soil characteristics and topography, and exports these as a new GIS layer.

Acceptable Levels: The Ultimate Acceptable level is the proportion of the area of agricultural fields (annual cropland, rotated cropland, permanent hay) that could theoretically support the practice in question. For cover crops and conservation tillage, this is equal to the area of land in annual crops (annual cropland plus one-half of the rotated corn-hay land use – this reflects an even rotation of corn and hay, which could be adjusted), and for manure injection it is both annual cropland and hayland (i.e. 100% of agricultural fields. The short-term acceptable levels are calculated by taking the areas noted in OFA and calculating the percent of the total area that these targets represent.

Expected Reductions: Expected reductions are calculated following the general method described in the text (box 2).

Cost data: Costs to achieve the expected reductions are calculated by multiplying the difference in acreage between the current state and the ultimate goal by the average payment per acre by VT AAFM across all FAP financial programs ($21), which was calculated by dividing the total amount paid to landowners for 2011 (reported in the 2011 ERP Annual Report) by the total acreage enrolled for that year. As discussed in the text, this (intentionally) does not include program administration costs (see Methods: Total Cost and Cost-Effectiveness).

% of agricultural land managed under an NMP

Current State: This number was taken from a recent survey by UVM Extension which surveyed dairy farms in Vermont (Darby et al. 2013). The survey requested information including total acreage managed by the farm (including rented or leased land), and whether the farmer had an actively maintained NMP (updated in the last 3 years). Those farms with actively maintained NMPs represented just under 65% of the land base. For the Quebec portion of the Missisquoi all farms are managed under NMPs, and so the higher percentage reflects an area-weighted ratio.

It is worth noting here that other data collected as part of the same survey indicated that in any particular year, producers apply the NMP recommendations to only 75% of their acreage. Survey responses identified poor weather as the primary cause for the less than 100% compliance. If that's true, then reaching 100% compliance in every year is impossible since the cause is a random occurrence.

Acceptable Levels: The long term-level was set by the AMWG in early meetings. There are no short-term levels specified in OFA for Vermont, and for New York, no data are currently available for acreage currently managed under NMPs, although a target is stated in OFA. The short-term goal is therefore arbitrary.

Expected Reductions: The reduction rate from Gitau et al. (2005) is applied to only 75% of the load from fields, to reflect the lower rate of use as noted above. Otherwise, the method is the same.

Cost estimates: There are no clear cost data for this indicator.

% farms with structural BMPs:

Current State: Values for Vermont are from the VT AAFM BMP database, delivered 4/2012. Practices were identified using selections from the "TPC title" field. For New York, numbers of practices on farms noted as "CAFO" were taken from the 2010 Ag BMP reporting project done by CWICNY and NYS Department of Agriculture & Markets (Snell and Brower 2011). Numbers of practices were aggregated to HUC 8 due to inconsistent use of the newer HUC 12 and older HUC 14 codes (on the Vermont side). Though the records in the Vermont AAFM database contained farm size information, the data we had for the total numbers of farms in each watershed only detailed the number of Medium and Large Farms (or Medium and Large CAFOs). Therefore, we only report numbers of practices for these regulated farms.

For the Vermont database, "Manure storage" in the indicator table includes practice records with the following TPC titles: Waste Storage Structure, Waste Storage Pond, Waste Transfer, Concrete Stacking Pad. "Silage Treatment" includes practice records with TPC title Waste Treatment – Silage. "Barnyard Runoff" includes practice records with TPC Titles: Barnyard Runoff Treatment, Roof Runoff, Diversion, Heavy Use area protection, Structure for Water Control. Milkhouse waste treatment includes practices with the following TPC titles: Milkhouse Wastewater Treatment, Milkhouse Wastewater Transfer, Milkhouse Wastewater infiltration area, Waste Treatment – Milkhouse, Wastewater infiltration area).

For the Vermont side, it should be noted here that there was no effective method to understand what numbers of practices are adequate for a particular farm from the database. In some cases, a single farm has cost-shared multiple manure pits, for example, and in some cases, a farm may have the necessary structures but not cost-shared them (a clear example is the single LFO in the Winooski watershed – that farm may have not cost-shared any of the practices, but it likely has structures necessary to manage waste.) It is therefore difficult to know whether the estimates are over- or underestimates. The best interpretation of these data is to compare the relative numbers of practices to farms to see where the emphasis has been (e.g. manure storage and barnyard runoff).

Acceptable Levels: In OFA 2010, New York and Vermont made commitments to ensure that all regulated farms (Medium and Large CAFOs) have these structures in place. The short-term acceptable level is therefore the same as the Ultimate acceptable level.

Expected Reductions: Manure storage and Barnyard runoff practice efficiencies were reported in Gitau et al. (2005) as percentages. The NY Ag BMP report provided rates for silage leachate and milkhouse waste treatment on an animal unit basis (Snell and Brower 2011). From the UVM Extension survey, we calculated a median AU/acre based on animal data provided by survey respondents, and also calculated a median AU for each farm size (Darby et al. 2013). To calculate reductions, the animal unit-based reductions were multiplied by the animal unit estimates per farm, and then multiplied by the remaining number of farms requiring that practice.

Cost Data: Cost data come from the Vermont State Act 78 Report of 2009, which estimated the need for these sorts of practices across the state. In addition to estimating numbers of farms in need of these practices, that report estimated a per-farm cost, or a per-AU cost (depending on the practice). Where per-farm costs were given, that cost was multiplied by the number of farms in need of that practice according to this analysis, and where per-AU costs were given, the median AU value for the relevant farm size was used.

% of inspections identifying violations:

Current State: Data from 2011 VT ERP Annual Report. Missisquoi data are from Québec MDDEFP. This is a simple average of the Vermont rate (5%) and the rate for Missisquoi inspections (13%), since there is not enough information to calculate a more appropriate ratio based on inspection numbers in the Missisquoi watershed.

Acceptable Levels: There are no short-term goals stated in OFA. The AMWG set the Ultimate Level.

Expected Reductions: For this project, we have given all inspection programs no reduction value. The logic is that the effectiveness we report and use in calculations assumes full compliance. Providing an additional reduction value for the inspections would double-count reductions. The role of inspection programs is to ensure that the level of reduction reaches its potential. One application of these data may be to subtract a proportion of the expected reductions according to a function of the non-compliance rate.

Developed Lands:

% of permitted construction stormwater sites in compliance:

Current State: Values taken from the Vermont ERP 2011 Annual report. There are no similar data available for Quebec or New York.

Acceptable Levels: No targets are set for this indicator in OFA, so there is no short-term acceptable level. The AMWG set the Ultimate Level.

Expected Reductions: See discussion above, in the Agricultural Lands section, about reduction effectiveness of inspection programs.

% of permitted operational stormwater sites in compliance:

Current State: Values taken from the Vermont ERP 2011 Annual report. There are no similar data available for Quebec or New York.

Acceptable Levels: No targets are set for this indicator in OFA, so there is no short-term acceptable level. The AMWG set the Ultimate Level.

Expected Reductions: See discussion above, in the Agricultural Lands section, about reduction effectiveness of inspection programs.

% impervious under stormwater management:

Current State: Acreages of impervious surface with stormwater permit provided by VT DEC from their stormwater database, and by NYS DEC. Acreages for Vermont were summarized by VT DEC Tactical Basin boundaries. Regarding the data from VT DEC: "Approximately 5.4% of the records are missing impervious acreage in the database. Impervious surfaces are generally not tracked for MSGP or Construction permits, so they were not included in this analysis." Therefore, this is an underestimate, but probably not significantly. Because of data entry issues with permit locations in their database, New York acreages were aggregated by town, and then summarized by HUC 8 basin. The total permitted impervious acreage for each basin (VT and NY) was divided by the area of impervious surface in each HUC 8 basin derived from the UVM SAL impervious area mapping effort (2011 imagery, NDVI + OBIA approach) (O'Neil-Dunne 2013).

A second possible method for estimating this is to use the total number of acres of impervious surface within regulatory boundaries (MS4 and stormwater impaired watersheds, where applicable). The values obtained from this method are roughly similar to the values from the above method that we used. There is a large degree of overlap (but less than 100%) between the two methods, so they can't be combined or substituted directly.

Acceptable Levels: There are no targets set for this value in OFA. The Ultimate level was set by the AMWG, but should probably be revised downward to reflect what is necessary to manage (i.e. estimates of the difference between total impervious areas [TIA] vs. effective impervious area [EIA]) and what is possible to manage in terms of considerations about the feasibility of on-site treatment.

Expected Reductions: Reduction values are from appendices D and E of the Center for Watershed Protection's Urban Subwatershed Restoration Manual 3 (available for free from www.cwp.org), which provides data on the effectiveness and costs of retrofitting stormwater management practices. The data in this report are compiled from several wide-reaching literature reviews, so these data incorporate a large number of studies.

Effectiveness values used in the Indicator Table are median effectiveness values for phosphorus across all types of practices included, as it is likely that in a large-scale effort to retrofit practices into the Lake Champlain Basin, a wide mixture of practices would be used.

Cost Estimates: Cost estimates used here are median costs to treat an acre of impervious surface, across all practice types. As mentioned above, the first estimate includes only base construction costs and design and engineering, not annual O&M, or land acquisition. The D&E costs are 32% of base construction costs, and include project management, design, permitting, landscaping, and erosion and sediment control during the construction phase. The second cost estimate uses annual maintenance costs from an estimate for retrofitting the Puget Sound urban areas, which is averaged across practice types.

Number of CSOs by town:

Current State: Numbers of current and recently abated Combined Sewer Outfalls[5] (CSOs) were given by VT DEC and NYS DEC.

Acceptable Levels: No targets are set for this indicator in OFA for the Vermont portion of the basin, though New York has committed to eliminating 50% of their outfalls by 2020.

Expected Reductions: To calculate reductions possible from eliminating CSO events, we required estimates of how much loading occurs from CSOs, and assumed that by eliminating outfalls, loading from CSO events would be essentially eliminated (though see Heath et al. 2004). Estimates of loading were developed by obtaining an estimate of the (1) number of overflow events per year per facility (from VT ANR Wastewater Division, reporting number of overflow events 2007-2011), (2) an estimate of the volume of overflow per event (from several monthly VT ANR Wastewater Overflow reports in 2011, which estimated volumes for some of the events: the range of volumes was 8,000 gal to 830,000 gal. The median of reported volumes was used, which was 190,000 gal per event), and (3) an estimate of the concentration of CSO effluent. Because the majority of the VT ANR overflow reports described having discharged "untreated sewage", an estimate of the influent TP concentration from the Middlebury WWTF was used. Those data are from Paul Klebs, of Aqua-Aerobic Systems Inc., which collected data on influent concentrations as part of a

[5] We use "outfall" in reference to the outfall pipe where combined sewer systems are discharged to a stream, and "overflow" to describe events when such a discharge occurs. In abbreviation, "CSO" refers to the outfalls, and "CSO events" refers to overflow events.

study of the efficiency of the Middlebury wastewater treatment system (Klebs 2008). Data were collected in 8 of the 12 months in 2002, and influent concentration averaged 17.8 mg/L TP. In wet-weather events (i.e. when CSO events most frequently occur), this concentration overestimates the true concentration (due to a dilution effect), which means that the estimates of reductions are probably optimistic.

Cost Data: Cost estimates of CSO elimination came primarily from the EPA report to Congress on CSOs and associated documents which provided cost estimates per acre treated by a CSO system, and estimates of cost per foot of CSO pipe eliminated (EPA 1999a). Estimates for the area of impervious surface in "downtown" areas of each town with CSOs were developed by calculating the area within a polygon surrounding the densely populated areas of each town where CSOs still exist. Because it's unlikely that the entirety of these areas is served by a combined system, these acreages were then multiplied by a reduction rate that was an estimate of the area actually served by combined sewers. The Burlington stormwater department reported these data in a 2008 report, which estimated that 60% of Burlington area was served by CSO. This rate was applied to all 10 towns with CSOs, and the EPA estimate of cost per acre was used. These rates should be adjusted in the future if better estimates of any of the input data are developed.

Urban Tree Canopy %:

Current State: Vermont Forest, Parks, & Recreation conducted an urban tree canopy assessment within "urban" land use zones, which uses E911 housing density to estimate parcel sizes. "Urban" zones are those where housing parcels are less than 5 acres in size. Urban tree canopy (UTC) percent was assessed in those zones. For this analysis, the UTC layer was overlaid with HUC 12 watershed boundaries, and UTC polygons were split and reassigned to the HUC 12 in which they reside. The HUC 12 received a tree canopy percentage that represents the area-weighted mean percentage for each of the UTC polygons in that watershed. The number reported here is best interpreted as the average tree canopy across all urban land use zones in the watershed.

Acceptable Levels: The Ultimate Level for this indicator has been set by Vermont's 2010 Forest Resources Plan (http://www.vtfpr.org/htm/documents/VT%20Forest%20Resources%20Plan.pdf).

Expected Reductions: There are currently no data available to estimate phosphorus reductions based on increasing UTC cover.

Cost Data: Similarly, there are no existing data for estimating cost of increasing UTC.

Rural Lands/Backroads:

% logging jobs causing sediment to enter streams:

Current State: USFS and the Northeastern Area Association of State Foresters (www.wetpartnership.org) conducted inspections on 94 sampling units in Vermont in 2004. These data are based on what the document identifies as "opportunities to observe" erosion, of which there are 5 per site (470 observations). Proportions of those "opportunities" therefore correspond roughly to site-level proportions.

Acceptable Levels: No short-term acceptable level has been set for this indicator.

Expected Reductions: The connection between upper watershed sediment loading and end-of-tributary phosphorus loading has not been articulated or quantified in a way that is applicable to this project. There are a wide variety of data on upper watershed sediment loading and downstream effects of various kinds, but phosphorus loadings have not been as clearly documented in this context.

Cost Data: Because there are a variety of management initiatives that pertain to this indicator, there is no clear way of calculating cost data for this indicator.

<u>% of towns participating in Better Backroads Program (or equivalent)</u>:

Current State: Data for Vermont are from taken from the ERP 2011 Annual Report, and specific to the Lake Champlain Basin as a whole. Data for New York are from the Lake Champlain - Lake George Regional Planning Board 2012 report (LCLGRPB 2012).

Acceptable Levels: There are no short-term targets set for this indicator. The AMWG set the long-term target.

Expected Reductions: See above, in the Agricultural Lands section, for reduction effectiveness of inspections and basic program involvement.

Cost Data: There were no clear cost data for "participation" in the BBR program that did not include administration costs.

<u>% of towns having completed erosion needs inventories and capital budget plans</u>:

Current State: Data for Vermont are taken from the VT ERP 2011 Annual Report, and specific to the Lake Champlain Basin as a whole. Data for New York are from the Lake Champlain - Lake George Regional Planning Board 2012 report to the LCBP. This report detailed capital needs by town, which addresses the intent for this indicator.

Acceptable Levels: There are no short-term targets set for this indicator. The AMWG set the long term target.

Expected Reductions: See above, in the Agricultural Lands section, for reduction effectiveness of inspections and basic program involvement.

Cost Data: There were no clear cost data for "participation" in the BBR program that did not include administration costs.

<u>% of priority erosion control projects identified in road erosion needs inventories that are completed</u>:

Current State: Data for Vermont are from taken from the ERP 2011 Annual Report, and are specific to the Lake Champlain Basin as a whole. It is unclear whether any of the projects identified in the LCLGRPB report have been completed.

Acceptable Levels: There are no short-term targets set for this indicator. The AMWG set the long-term target.

Expected Reductions: There are no good data for estimating the effect on phosphorus loss of managing roadside erosion through the sort of erosion control practices this indicator describes. Beverley Wemple (UVM) is currently in the second phase of a project to evaluate common BMPs for unpaved road maintenance (Wemple 2013). Until these data are published, the reduction rate data used here should be used as a stand-in only. The reduction estimate used here refers to total sediment, NOT total phosphorus, and at 65%, is likely an overestimate for total phosphorus, which means that the expected reductions should probably be lower, and the cost per kg of phosphorus should be higher.

Cost Data: Cost data are from the LCLGRPB report. The estimated cost to remedy each of the erosion problems they documented. We calculated the average cost of this group of projects (n=319, with a total estimated cost of slightly more than $1.7M), which ranged widely from hundreds to tens of thousands of dollars. We then estimated how many of these erosion control projects had not been completed for the Vermont side (from the ERP 2011 Annual Report) and then applied the average cost per project to the number of projects remaining. One caveat of note is that many of the erosion projects for which costs were estimated in the LCLGRPB project involved light grading and hydro-seeding, which is very inexpensive, but probably confers significantly less P reduction potential when compared to stone-lined ditches, for example. Therefore, this estimate of project cost may be an underestimate for the phosphorus reduction rate, despite being reasonably accurate across all project types.

River, Floodplain, and Wetland Conservation and Restoration:

% of towns adopted standards in accordance with Act 110:

Current State: List of towns having adopted Act 110 codes & standards, delivered from VT DEC, last updated 7/2/2012.

Acceptable Levels: No targets exist for this indicator in OFA. The AMWG set the Ultimate goal.

Expected Reductions: There are no existing data that describes a relationship between standards for riparian area development and construction and phosphorus reductions downstream.

Cost Data: There are no cost data for this indicator that do not include program administration.

% of towns with adopted municipal Fluvial Erosion Hazard ordinances:

Current State: List of towns with FEH ordinances taken from the VT ERP 2011 Annual Report.

Acceptable Levels: The targets for this indicator in OFA are not directly translatable into the structure for this table. The AMWG set the Ultimate goal.

Expected Reductions: There is no existing data that describes a relationship between town-level ordinances and phosphorus reductions downstream.

Cost Data: There are no cost data for this indicator that do not include program administration.

Wastewater:

% of facilities meeting their relevant regulatory allocations:

Current State: Wastewater Treatment Facility (WWTF) load limits and actual loads were delivered by Eric Smeltzer (VTDEC) in October of 2012. The tables document TMDL allocation for each facility and its actual load, enabling easy calculation of how many facilities exceeded their limit over a three-year average. Entries were then grouped by watershed to calculate a percentage of facilities within the basin that meet their target.

Acceptable Levels: Acceptable levels were set by the AMWG during the initial stages of the indicator development.

Expected Reductions: In this case, expected reductions equaled the difference between the 3-year average load and the regulatory limit set for the facility. These differences were summed within watersheds.

Cost Data: No cost data for bringing treatment facilities up to 100% compliance was found.

% of facilities having approved Spill Prevention Plans:

Current State: Status reports for the approval of WWTF spill prevention plans were provided by Eric Smeltzer late September 2012. The figures provided here are simple tallies within watersheds.

Acceptable Levels: Acceptable levels were set by the AMWG during the initial stages of the indicator development.

Expected Reductions: Expected reductions are impossible to calculate for this indicator, because loading estimates from WWTF spills is not available.

Cost Data: No cost data for bringing treatment facilities up to 100% compliance was available.

Ecosystem Response Indicators:

Median Animal Units per acre:

This number was from a recent survey by UVM Extension that surveyed dairy farms in Vermont. Among the information collected in the survey was total acreage managed by the farm (including rented or leased land), and numbers of animals. From these data we calculated a median animal unit value per acre. Survey respondents also reported their county, and the value here by watershed is the area-weighted average of the county-level values for the counties within that watershed.

P loss from cropland, farmsteads, urban areas, and the road network:

Tetra Tech reported proportions of the total load from each land use category in the calibration report for their SWAT model. The proportions were applied to the total load they reported, to keep consistent with their modeling results. For the Grand Isle direct drainage area, land use areas within the watershed were multiplied by the mean loading rates (across all drainages) identified in the SWAT calibration report (Tetra Tech 2013a).

% land in annual crops, hay/pasture/lawn, and impervious surface:

Annual crops: Land use data from Tetra Tech SWAT modeling effort was used to estimate the area in each HUC 8 level watershed. "Annual cropland" is the sum of corn, soy, etc. plus the generic cropland category, plus ½ of the crop-hay rotation land use, which assumes that the rotations are of equal time (e.g. 4 years corn, 4 years hay). This could be adjusted to reflect a more dominant rotation (which, for example, may be 6 corn/4 hay, raising the proportion in corn over the long term).

Hay pasture lawn: the grass/hay/pasture category was constructed from the Tetra Tech land use layer (Tetra Tech 2013a), summing the area of herbaceous (71), pasture (81), and hay (87), and adding half of the corn-hay rotation class (to complement the annual crops area), and adding 80% of the Developed – open category. This last addition reflects the "lawn" portion, which is generally counted in either the herbaceous category where it's in a rural context, or in the open development when in an urban context. No more than 20% of the Developed Open category is in impervious surface, and by extension, no less than 80% is "open", or lawn.

Impervious surface: estimates from UVM SAL's impervious surface analysis were summed for each HUC 8, and divided by the total area of the watershed (shape_area field minus "water") (O'Neil-Dunne 2013).

<u>% River miles in Channel Evolution Stage II or III:</u>

These data are taken from the VT River Management Program's Phase 2 Stream Geomorphic Assessment data sheets. We downloaded all datasheets for watersheds in the Lake Champlain Basin, and tallied those reaches in each channel evolution stage. Similar data were not available for New York or Québec.

<u>5 yr. avg Wastewater load</u>:

Running average over the last 5 years (2007-2011). Tables delivered by Eric Smelzter 10/1/12, updated through 2011.

<u>5-year avg. NP load (2007-2011):</u>

This is estimated by subtracting the yearly wasteload from the yearly estimate of the total load (flux) obtained from the WRTDS procedure. The values are then averaged. For basins that include more than one drainage, these averaged values are summed to get the "whole basin" average non-point load.

<u>5-year avg. total load:</u>

This is estimated using the WRTDS method, using the program defaults, which seem to perform well, for most tributaries. Flux bias statistics were similar to Medalie (2013).

<u>DP:TP flux</u>:

This is calculated by summing the estimated fluxes for each 3-month season (broken by water year, such that fall is Oct. 1 – Dec.31, winter is Jan. 1-March 31, spring is April 1 – June 30, and summer is July 1 – September 30), and then calculating the ratio of the season-specific fluxes per year.

APPENDIX C – BAYES NETWORK MODEL DATA

Table C - 1. Watershed-specific data used to populate the Bayesian network

Variable	Winooski	Missisquoi	Lamoille	Otter Creek
Total Watershed Area (ha)	283604	154276	188014	224019
Water area (ha)	2455	753	2037	1830
Urban area (ha)	9282	2841	3344	5033
Opendev area (ha)	22344	4188	11297	12341
Forest area (ha)	222241	109142	146728	151869
GrassPastHay area (ha)	17258	23846	14650	34992
Crops area (ha)	7782	9490	7642	8876
Wetlands area (ha)	2242	4016	2315	9078
Total impervious surface (ha)	6837	2398	3133	4173
Urban area within MS4 (ha)	2955	0.01	278	451
Opendev area within MS4 (ha)	1650	0.01	252	172
MS4 road impervious LU class 20 (ha)	304	0.01	24	38
MS4 other impervious LU class 20 (ha)	863	0.01	52	119
Non-MS4 road impervious LU class 20 (ha)	639	261	259	454
Non-MS4 other impervious LU class 20 (ha)	920	326	391	757
MS4 road impervious LU class 21 (ha)	83	0.01	12	4
MS4 other impervious LU class 21 (ha)	134	0.01	24	14
Non-MS4 road impervious LU class 21 (ha)	436	180	209	251
Non-MS4 other impervious LU class 21 (ha)	375	136	202	183
Non-MS4 other impervious LU class 40 (ha)	1011	359	608	496
Non-MS4 road impervious LU class 40 (ha)	291	127	184	197
Proportion backroad area in LU class 21	0.370	0.256	0.288	0.329
Proportion backroad area in LU class 40	0.405	0.346	0.409	0.317
Proportion backroad area in LU class 81	0.097	0.227	0.145	0.255
Proportion backroad area in LU class 20	0.082	0.106	0.089	0.051
Total backroad area (ha)	898	425	578	465
Proportion of permanent hay in LU class 81	0.748	0.810	0.741	0.855
Proportion of pasture in LU class 81	0.169	0.172	0.208	0.125
SFO farmstead area (ha)	8185	11409	7470	14966
Cover crop feasibility area (ha)	6656	8365	6359	7802
Reduced Tillage feasibility area (ha)	3328	2117	2503	4738
Estimated area with tile drainage (ha)	778	3796	764	1775
Manure Injection (crops) feasibility area (ha)	7392	9064	7263	8174
Manure Injection (hay) feasibility area (ha)	12909	19315	10856	29918
Riparian buffer feasibility (km)	84	314	325	405
Number of SFO barnyards	309	415	263	394
Wetland area under LU class Pasture & Hay	583.2	896.67	535.86	3139.56
Wetland area under LU class Cropland	340.38	635.58	562.59	947.43
Target NPS tributary load (kg/yr)	109287	36450	38572	80573
Target total tributary load (kg/yr)	118816	38076	41504	89988

Table C - 2. Phosphorus reduction efficiencies used in the Bayes network model. All values are proportions.

Management Practice	Minimum	Mode	Maximum
Backroad BMPs	0.585	0.65	0.715
BMP Design Size - 0.25 in (MS4, Road)	0.3	0.445	0.54
BMP Design Size - 0.25 in (Non-MS4, Road)	0.3	0.445	0.54
BMP Design Size - 0.25 in (MS4, Other)	0.3	0.445	0.54
BMP Design Size - 0.25 in (Non-MS4, Other)	0.3	0.445	0.54
BMP Design Size - 0.5 in (MS4, Road)	0.42	0.525	0.77
BMP Design Size - 0.5 in (Non-MS4, Road)	0.42	0.525	0.77
BMP Design Size - 0.5 in (MS4, Other)	0.42	0.525	0.77
BMP Design Size - 0.5 in (Non-MS4, Other)	0.42	0.525	0.77
BMP Design Size - 0.9 in (MS4, Road)	0.34	0.545	0.92
BMP Design Size - 0.9 in (Non-MS4, Road)	0.34	0.545	0.92
BMP Design Size - 0.9 in (MS4, Other)	0.34	0.545	0.92
BMP Design Size - 0.9 in (Non-MS4, Other)	0.34	0.545	0.93
BMP Design Size - 1.5 in (MS4, Road)	0.56	0.745	0.98
BMP Design Size - 1.5 in (Non-MS4, Road)	0.56	0.745	0.98
BMP Design Size - 1.5 in (MS4, Other)	0.56	0.745	0.98
BMP Design Size - 1.5 in (Non-MS4, Other	0.56	0.745	0.98
BMP Design Size - 2.0 in (MS4, Road)	0.65	0.775	0.99
BMP Design Size - 2.0 in (Non-MS4, Road)	0.65	0.775	0.99
BMP Design Size - 2.0 in (MS4, Other)	0.65	0.775	0.99
BMP Design Size - 2.0 in (Non-MS4, Other)	0.65	0.775	0.99
Cover Crops	0.11	0.46	0.69
Livestock Exclusion	0.58	0.67	0.76
Manure Injection (Crops)	-0.47	0.08	0.63
Manure Injection (Hay)	0.34	0.55	0.65
Reduced Tillage	-0.33	0.27	0.82
T Erosion Standard	0.20	0.24	0.27
Tile Drainage Treatment	0.32	0.50	0.87
Tile Drainage Contribution	0.06	0.26	0.48
Riparian Buffer	0.21	0.55	0.89
Barnyard Runoff Separation	0.72	0.80	0.88

Table C - 3. Cost values used in the Bayes network model. All values are in units of dollars per hectare per year.

	Minimum	Mode	Maximum
Backroad BMP	5165.1	5739	6312.9
BMP Design Size - 0.25 in (MS4, Road)	1037	2047	2592
BMP Design Size - 0.25 in (Non-MS4, Road)	1037	2047	2592
BMP Design Size - 0.25 in (MS4, Other)	1037	2047	2592
BMP Design Size - 0.25 in (Non-MS4, Other)	1037	2047	2592
BMP Design Size - 0.5 in (MS4, Road)	1555	4147	5184
BMP Design Size - 0.5 in (Non-MS4, Road)	1555	4147	5184
BMP Design Size - 0.5 in (MS4, Other)	1555	4147	5184
BMP Design Size - 0.5 in (Non-MS4, Other)	1555	4147	5184
BMP Design Size - 0.9 in (MS4, Road)	2799	7465	9331
BMP Design Size - 0.9 in (Non-MS4, Road)	2799	7465	9331
BMP Design Size - 0.9 in (MS4, Other)	2799	7465	9331
BMP Design Size - 0.9 in (Non-MS4, Other)	2799	7465	9331
BMP Design Size - 1.5 in (MS4, Road)	4666	12442	15552
BMP Design Size - 1.5 in (Non-MS4, Road)	4666	12442	15552
BMP Design Size - 1.5 in (MS4, Other)	4666	12442	15552
BMP Design Size - 1.5 in (Non-MS4, Other	4666	12442	15552
BMP Design Size - 2.0 in (MS4, Road)	6221	16589	20736
BMP Design Size - 2.0 in (Non-MS4, Road)	6221	16589	20736
BMP Design Size - 2.0 in (MS4, Other)	6221	16589	20736
BMP Design Size - 2.0 in (Non-MS4, Other)	6221	16589	20736
Cover Crops	91	188	235
Livestock Exclusion	157	185	213
Manure Injection (Crops)	68	76	83
Manure Injection (Hay)	68	76	83
Reduced Tillage	37	40	104
T Erosion Standard	46	61	76
Tile Drainage Treatment	457	486	517
Riparian Buffer	1322	1561	1799
Barnyard Runoff Separation	2552	2835	3119